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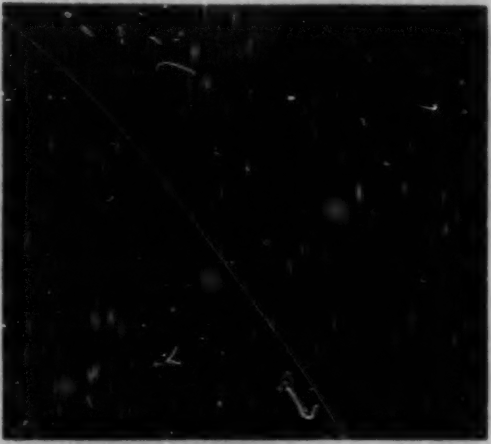

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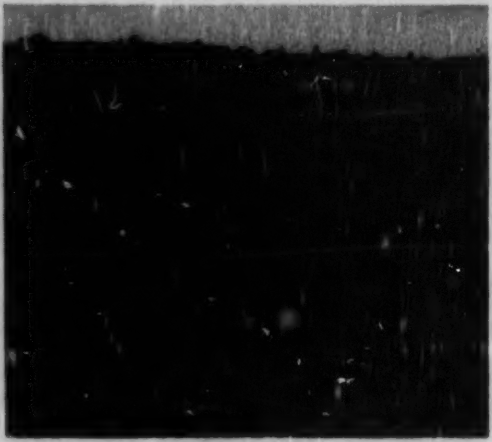

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The ~~proceedings of a~~ 1975 conference was sponsored by the Virginia Water Resources Research Center in cooperation with the U.S. Office of Water Research and Technology, EPA Regions III and IV, and Virginia Polytechnic Institute and State University. Purposes included the transfer of research findings and methodologies to user groups, and researcher-user discussion of knowledge gaps and research needs. Includes 21 major papers, extensively footnoted and with numerous figures and tables. Five papers provide overview and state-of-the-arts review. Thirteen papers give specific reports on research, case studies, and examples in the four principal source areas from which water pollution originates: (1) agricultural and rural; (2) forestry; (3) urban and construction, and (4) mining and mineral extraction. A concluding section includes three papers that discuss control remedies, policies, and programs. Throughout, a common point of reference is the Federal Water Pollution Control Act Amendments of 1972, and in particular Section 304(e) of that legislation. An appendix contains a complete list of the names and addresses of conference registrants and speakers.

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*Pollutant Identification, Runoff, Turbidity, Runoff Coefficient, *Urban Runoff, ~~Acid~~ *Mine Drainage, *Water pollution sources,
Erosion, Combined Sewers, Sediments, Coliforms, Pesticides, ~~Non-Point~~
*Agricultural Runoff, Farm wastes, Nutrients, Agricultural Chemicals, *Strip mining, ~~Mined~~ Land Reclamation, Heavy metals, Southeastern U.S., *Sediment yield, Construction, Biochemical oxygen demand, Forest management.

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~~Surface mining~~
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Proceedings of a Southeastern Regional Conference
Conducted on May 1 and 2, 1975, in Blacksburg, Virginia
at Virginia Polytechnic Institute and State University

Sponsored by

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The Virginia Water Resources Research Center is a federal-state partnership agency established under the Water Resources Research Act of 1964. The Act provided for establishing in each state an agency which would "stimulate, provide for, and supplement present programs for the conduct of research, investigations, experiments, and the training of scientists in the fields of water and resources which affect water." In fulfilling this responsibility for Virginia, the Center serves three primary functions: (1) It surveys the state's water resource problems; (2) it sponsors and administers water resources research, and (3) it collects and distributes information on water resources. Principal funding for the Center's activities in these areas is provided by the Office of Water Research and Technology, U.S. Department of the Interior.

Contents

PREFACE

1. Peter M. Ashton

KEYNOTE ADDRESS

5. "Balance and Perspective in Non-Point Source Water Pollution Control Programs"
Warren A. Hall

OVERVIEW, IMPORTANCE, STATE OF THE ARTS

11. "An Overview of Agricultural and Rural Non-Point Water Pollution in the Southeastern United States"
Victor J. Kilmer

29. "Southeastern Forests and the Problem of Non-Point Sources of Water Pollution"
James E. Douglass

45. "An Overview of Non-Point Water Pollution from the Urban-Suburban Arena"
Harold P. Guy

67. "Non-Point Pollution from Mining and Mineral Extraction"
Ronald D. Hill

RESEARCH REPORTS, CASE STUDIES, EXAMPLES

Agricultural and Rural

85. "Methodology for Assessing Rural Non-Point Source Pollution"
F. J. Humenik, F. Koehler, M. R. Overcash

101. "Animal Waste and Non-Point Pollution"
Eldridge R. Collins, Jr.

115. "Agricultural Chemicals and Their Importance as a Non-Point Source of Water Pollution"
J. B. Weber

131. "A Quantitative Study of the Nutrient, Sediment, and Coliform Bacterial Constituents of Water Runoff from the Rhode River Watershed"
David L. Correll, J. W. Pierce, Maria A. Faust

Forestry

145. "Harvesting Southern Forests: A Threat to Water Quality?"
S. J. Ursic

153. "Erosion and Sedimentation as Related to Intensive Timber Management in the Southeast"
Wade L. Nutter

157. "Forest Management Practices as Related to Nutrient Leaching and Water Quality"
Edward S. Corbett, James A. Lynch, William E. Sopper

Urban Contributions

175. "Evaluation of Sediment Yield Due to Housing Construction: A Case Study"
Chin Y. Kuo

195. "Characterization of Urban Land Runoff"
Newton V. Colston

217. "Heavy Metals in Urban Runoff"
William G. Wilber, Joseph V. Hunter

Mining and Mineral Extraction

239. "Stream Rehabilitation through Control of Non-Point Sources of Acid Mine Drainage"
Edwin E. Herricks, John Cairns, Jr.

257. "Sediment Control During Mining and Mine Reclamation"
Michael A. Nawrocki

267. "Kentucky's Program for Improved Surface Mining Technology: Administration and Reduction of Adverse Environmental Impact"
Robert E. Nickel

CONTROL REMEDIES, POLICIES, AND PROGRAMS

279. "Non-Point Water Pollution: Federal and State Perspectives"
John R. Churchill

287. "Private Control Measures for Non-Point Water Pollution"
William E. Cox

297. "Areawide Waste Treatment Management of Non-Point Pollution"
Paul Edward Barker

APPENDIX

307. Conference Registration List

Preface

The Federal Water Pollution Control Act Amendments of 1972 established as "the national goal that the discharge of pollutants into the navigable waters be eliminated by 1985." Thereby, pollution abatement became elevated to a national objective of highest priority. Since passage of the 1972 Act, point sources of water pollution have received much attention. Substantial progress has been made in development and application of both point source control technology and the administrative, regulatory, and enforcement institutions required for point pollution minimization.

It is becoming increasingly clear, however, that point source control must be considered as only one part of the total water pollution abatement effort. Even as programs and technological advances show significant progress with respect to the point source problem, it is evident that no amount of investment, whether public or private or both, will be sufficient to ensure attainment of the 1985 zero discharge goal without first coming to grips with another aspect of water pollution--the non-point, or diffuse, source problem.

The 1972 Act properly recognized the existence of this dimension of water pollution control. Section 304(e) of the Act requires the Administrator of the Environmental Protection Agency to issue information including "(1) guidelines for identifying and evaluating the nature and extent of non-point sources of pollutants, and (2) processes, procedures, and methods to control" non-point sources of water pollution. A number of studies have been undertaken and several documents have been issued by EPA, their contractors, and others in response to the directives contained in this section of the 1972 Act. Much more research is presently underway. It is now firmly recognized that the quality of the nation's water resources is a combined result of processes attributed both to man's activities and to natural processes. The latter tend to be diffuse, or of a non-point nature, and are not amenable to direct treatment. Some of the former may be discrete and treatable; others may similarly be characterized as non-point. In general, these non-point sources resulting from man's activities may be classified broadly into four categories following the wording of the 1972 Act. These are:

1. agricultural activities,
2. forest-related activities,
3. mining activities, and
4. construction activities.

While there remains substantial disagreement regarding the nature and extent of the pollution that falls under the heading of "non-point," there is universal acknowledgement that non-point sources are a major deterrent to the achievement of national water quality goals. Characterization of these non-point sources and formulation of plans for their amelioration will inevitably occupy the minds of research and development, planning, implementation, and enforcement authorities during the latter part of the seventies just as did point source issues during the early seventies and before.

The high costs of water pollution abatement dictate that relevant public policy and programs be soundly conceived. The pace of development in this area demands frequent scrutiny, stock-taking, and evaluation. Substantial research dollar amounts, both public and private, are involved; extensive publicly funded planning and management programs are underway. It is imperative that officials charged with carrying out these programs operate with the best, most current information available.

With these considerations in mind, the Virginia Water Resources Research Center, in May 1975, sponsored a comprehensive program covering the state of the arts of the four principal dimensions of non-point sources of water pollution listed above. Regional and national experts gathered at Virginia Tech in Blacksburg, Virginia, for two days to discuss and disseminate to a large and varied audience the latest findings, developments, and thinking on non-point pollution control technology, and management and enforcement programs. Some 21 significant papers were presented at this conference and are contained in this volume. It is hoped that the substance of these papers will represent a worthy addition to the growing literature on non-point sources of water pollution.

This conference was made possible through the financial support of a number of institutions in addition to the Virginia Water Resources Research Center. These included the Office of Water Research and Technology, U.S. Department of the Interior; the U.S. Environmental Protection Agency, Regions III and IV; and the Research and Extension Divisions of Virginia Polytechnic Institute and State University. We are most grateful to the support of these institutions. Recognition and thanks are also extended to the staff of the Virginia Water Resources Research Center for their various contributions and, in particular, to Dick Underwood for his editorial and publication assistance, Victoria Esarey for her manuscript typing and Sherri Carroll for her friendly assistance with numerous conference details.

Others could be singled out for recognition. However, the success of the conference is largely due to the contributions of speakers on the program and the exchange of information and ideas, both formal and informal, which they encouraged. The pages that follow are testimony to their competence.

Peter M. Ashton
Blacksburg, Virginia
June, 1975

Keynote Address

BALANCE AND PERSPECTIVE IN NON-POINT SOURCE WATER POLLUTION CONTROL PROGRAMS

Warren A. Hall
Elwood Mead Professor of Engineering

Colorado State University
Fort Collins, Colorado

In many respects this conference on non-point sources of water pollution is particularly timely. It was about a decade ago that the increasing sense of urgency for cleaning up our natural waters first resulted in a solid national commitment in dollars. The somewhat timid funding suggested in the water quality legislation in 1965 seems woefully inadequate in the light of today's assessment of the problem, but it did indeed back up the "please don't pollute" of prior years with an action program of sufficient size to serve notice that we meant business.

At the same time, we must acknowledge that the general view of stream pollution at that time was virtually synonomous with untreated municipal sewer outfalls, and the pathogenic component of pollution was the central concern.

In 10 short years, the public concept of what constitutes pollution has expanded exponentially. Hundreds of substances, considered quite innocuous by all but a few scientists a decade ago, are today presented by those who should know as constituting 100 public enemies, each of which appears to be number 1. Nitrate, phosphate, various heavy metals, radioactive elements, silt and sediment, chlorine, sodium, acid, many carcinogens, and many more types and forms of trace substances have been indicted in scientific journals and in the public press. Indeed, one expects at any moment to be informed dramatically that our drinking water is full of H_2O, and that H_2O in significant quantities has been found in the tissues of every single creature that has died during the past 10 years.

With all due respect to those producing the monthly crop of new pollutants, the most significant finding of the past decade is that for virtually every category of pollutant, those contributed from diffuse or non-point source origin appear to equal or exceed those from point sources, often by a substantial margin.

There are a number of reasons why this must be considered the most important observation of the decade. First, it informs us rather emphatically that, despite best intentions, the multi-billion dollar water clean up program now getting up a good head of steam isn't going to accomplish much of anything, except in a few very flagrant cases. Second, it invokes a sense of some urgency in organizing, funding, and executing a major program of research and development directed toward controlling these sources. Third, it may provide the opportunity to reassess our environmental protection situation and allow the

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introduction of better balance and sense of scheduling priority to our national water quality programs.

The latter is the topic of a paper I have been asked to prepare for the upcoming American Water Resources Association Conference at Rutgers University. Today, however, I would like to elaborate a bit on the first two questions.

I mix one glass of dirty, undrinkable water with a glass of pristine pure water, and all I get is two glasses of dirty, undrinkable water.

It really doesn't matter much whether the resultant glass you drink from contains a million pathogenic organisms per mililiter or only 500,000 of them. It doesn't make much difference to the fish whether the turbidity is 2 or 10,000 parts per million. Pick up half the litter scattered along a highway, what is left is still an eyesore. The change is difficult to observe.

Consider water pollution from the same perspective. Only when the resultant mixture is well below the tolerable threshold of the contaminating factors concerned will it pay to spend large sums of money to clear up half the source. Even then, when the cost of removing each additional increment of pollutant is increased dramatically as more and more is removed, there is a very strong suspicion that some of this money might better be spent looking at the other, untouched half of the source.

The present availability of such resources as technical design know-how, cash funds, bond market capacity, construction equipment and skills, skilled operators, and so on makes it quite impossible to wipe out every point source of pollution at one show. In fact, it can't be done in several decades. The General Accounting Office report of 1970 urged, rather fruitlessly, that a lot more would be accomplished in far less time and with considerably less money by adopting some rational basis for the job instead of the random, "we haven't got time to plan" approach inherent in both the law and its implementation.

We now are clearly faced with the rather sobering fact that diffuse sources in every major urban area observed were contributing as much or more of the same adverse constituents to streams, lakes, and estuaries as were the urban point sources. When we add the non-urban diffuse sources, it is clear that our great crusade against water pollution is mired in the mud long before it gets to the holy land.

Thus, we are faced with a serious threat to the attainment of our water quality goals. Before we can proceed, we will be required to solve some difficult problems for which we have no ready solutions on the shelves.

At the risk of sounding like a broken record, I must repeat a fact which I have frequently asserted (albeit speaking to the wall in most cases)--that in the final analysis problems are resolved only by the use of knowledge and understanding of the various physical, biological and social systems involved and a generous dose of pure guess. We call the combination of these two components of problem solving "judgment," but this nomenclature does not in any way change the percentage of guesswork.

Research, broadly defined, is the process by which we gain additional knowledge and understanding of any system or subsystem. If we increase this knowledge and understanding, we should be able to decrease the guesswork and increase the probability of producing good solutions to our problems.

However, this isn't the way we seem to go about solving problems when they require political decisions. Rather, we spend considerable amounts of time and energy going through four phases, only the last of which is apt to permit much progress.

First, we always ignore a new problem in hopes that it will go away. We knew about the dirty glass and the clean glass some time prior to 1970, but we have ignored it in virtually every bit of national policy and program legislation up to the present.

Second, we try to declare the problem illegal. We assert that it is herby prohibited. Even Walt Kelley's lampoon with his "non-breathers" club has not been able to shake us out of the false security of this second step. If it hasn't already been done, I suspect we will use current law to declare all non-point sources illegal.

At this stage, it becomes apparent that declaring pi to be equal to 3.0 by law doesn't make it so. Things are now becoming a bit sticky. The battle may be considered to have been won, but the enemy is still in the field and knocking on the gates of the city. So we invoke society's classical third step: we try to drown the problem in money.

When this doesn't work either (and it won't), we will recognize the facts of life and get on with the business of trying to use what little knowledge we have managed to put together to leaven a rather substantial batch of previous guesswork, in hopes of arriving at a solution. I can cite hundreds of cases where that leavening was hardly adequate to raise the issues, let alone solve the problem. At last we recognize the need for research. But now we are told, "This is a crisis. By the time you can do the research, we will have had to make a decision." So we go ahead and guess, and forget about research.

Of course, this doesn't really solve anything. Although some guesswork will always be required for the solution to any problem, it does not follow that we can expect guesswork that isn't backed by considerable knowledge and understanding to do much more than make the problem worse. In the case of non-point source pollution control, we are not going to get by with guessing. We need research, we need it urgently, and it had better be on target if the U.S. clean water program is to avoid resemblance to a Keystone Cop comedy.

Having said this much, it is essential that I go the next mile and point out to the research scientist that his work has to be focused on the problem. We can't afford to chase interesting sidelights. We need answers to the problem at hand. We cannot expect anyone to do anything more than guess if our science isn't relevent to the specific problems at hand. Furthermore, we need to start looking at the relative importance of the various factors we are concerned with. Not all materials reaching a stream are equally deleterious. Some materials have been given a lot of fanfare all out of proportion to their nature and probable hazard potential. There simply will not be enough research dollars to chase down the system history of every trace element which "might" cause problems.

The purpose of this conference is to present what we do know about non-point sources of pollution and what this might mean when it comes to remedial action. With the diversion of attention to the energy crisis today and the potential diversion of attention to the impending food crisis added to economic recession and inflation, it is very clear that we cannot afford to spend unlimited funds to clean up half the problem. It is time we gave priority attention to the second half--so that when they are ultimately mixed in the stream, we don't have a staggering bill to pay for two dirty, undrinkable glasses of water.

Overview, Importance, State of the Arts

AN OVERVIEW OF AGRICULTURAL AND RURAL NON-POINT WATER POLLUTION IN THE SOUTHEASTERN UNITED STATES

Victor J. Kilmer
Chief, Soils and Fertilizer Research Branch

Division of Agricultural Development
National Fertilizer Development Center
Tennessee Valley Authority
Muscle Shoals, Alabama

INTRODUCTION

In the United States we have had two distinct periods with opposite trends in the productivity of the land. The first period, in which the productivity of land declined, roughly paralleled the period of settlement which ended at the beginning of this century [Marshner, 1959]. During this era, demands for increased agricultural production were met primarily by bringing new land into cultivation. New land was plentiful, hence its use was exploitive. One need only read Edmund Ruffin's essay on calcareous manures, first published in 1852, to substantiate this fact [Ruffin, 1852]. According to Ruffin, the prevailing practice was to "take up new land, the best to be had, grow tobacco for three or four years, and then Indian corn as long as any will come. And in the end, if the soil is thoroughly impoverished, they begin again with a new piece and go through the rotation."

During the latter part of the 19th century, science and technology began to transform farming operations and, as the U.S. population increased, a parallel increase in demand for food resulted. Although U.S. population has nearly doubled since 1920, cropland has been reduced by about 20 million acres during the past 25 years; yet the yield of principal crops has increased two- to threefold in the 1950-1970 period [New York Times Encyclopedia Almanac, 1970]. Fertilizer use has also increased fourfold since 1950 [Harre, 1974], and pesticide use is up from 637 million to well over a billion pounds at present [Metcalf, 1971]. Fertilizers and pesticides are two of the major reasons that a farmer is able to feed and clothe 51 persons today, compared with 25 persons in 1960 and 4 persons in 1850.

However, our vastly increased agricultural production is not without its drawbacks. The increased use of chemicals always poses a threat to the environment unless care is taken to use these chemicals properly. And it is certain that our agricultural lands will be even more intensively cropped in the future if we are to maintain an abundant and economical supply of food for our increasing population. The main question becomes: Is maximum food and fiber production possible without intolerable water pollution?

This paper attempts to examine the present state of agricultural practices

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and production in the southeastern United States and their relationship to potential pollution problems. Such an analysis may suggest some reasonable answers to this question. But bear in mind this recent statement by Dr. W. F. Huey, Jr., Director of the Minnesota Agricultural Experiment Station: "The future currency of world trade will be food." It is not unreasonable to assume that more tradeoffs will be required in the future in the areas of food production and environmental quality.

GENERAL CONSIDERATIONS

Climatic and Hydrologic

For purposes of this discussion, the Southeastern states include Alabama, Florida, Georgia, Kentucky, Mississippi, North Carolina, South Carolina, Tennessee, and Virginia. Rainfall in this region ranges from 40-60 inches and the growing season is relatively long, spanning 160-320 days. Surface runoff varies from 10-40 inches and potential evapotranspiration from 30-50 inches [Water Atlas of U. S., 1973; Yearbook of Agriculture, 1941]. Precipitation generally exceeds evapotranspiration during the winter and early spring months. Thus the average annual potential percolation rate is high during this period, possibly approaching or exceeding 7 inches of water over most of the region.

Land Area and Cropland

The total land area of the southeastern states is about 263 million acres. About 51 million acres or 19% is cropland, contrasted with 56% in selected Midwest states [Table 1]. Vegetable production, along with cotton, soybean, and corn acreage, is shown in Table 2. These crops were selected because of their potential relationship to pollution hazards. The combined acreages of cotton, soybeans, and corn represent about 41% of the total cropland acreage in the southeastern states.

Soils

The soils of the southeastern states are not easily characterized in concise, quantitative terms. Except for rather isolated areas, such as the Black Belt soils in Alabama and Mississippi, the upland soils of the Southeast were developed under hardwood forests, while the soils of the Coastal Plain developed under coniferous vegetation. The parent material includes limestone, sandstone, shale, and igneous rocks, as well as unconsolidated Coastal Plain deposits, alluvium, and loess. The majority of the soils are acid, low in organic matter, nitrogen, and phosphorus. Many soils also have a low exchange and water-holding capacity. Their inherent fertility is thus quite low.

The low native fertility of extensive areas of non-agricultural and non-forested upland soil areas in the Southeast often results in a sparse vegetative cover. These soils are potentially erosion-prone. Leaching is the chief problem with light-textured soils typical of the Coastal Plain, where a flat topography prevails.

TABLE 1

Total Land Area and Cropland in the Southeastern United States and Selected Midwestern States.

Region	Total Land Area	Cropland	Cropland
	- - - - 1 x 10^6 Acres	- - - -	%
Southeast*	262.9	50.7	19
Range	--	--	11-26
Midwest**	273.5	150.1	56
Range	--	--	41-74

*Alabama, Florida, Georgia, Kentucky, Mississippi, North Carolina, South Carolina, Tennessee, and Virginia.
**Illinois, Indiana, Iowa, Kansas, Minnesota, Nebraska, and Ohio.

1973 Agricultural Statistics, USDA

Looking at the region as a whole, the climatic, hydrologic, and soil characteristics of the Southeast combine to favor the water transport of nutrients, pesticides, and sediment. However, many other factors influence the degree of pollution hazards involved. Among these factors are the extent and use-intensity of agricultural land, inputs of agricultural chemicals, other production practices, and degree of erosion control.

SOURCES AND POTENTIAL POLLUTION HAZARDS OF NUTRIENTS, ANIMAL WASTES, PESTICIDES, AND SEDIMENTS

Plant Nutrients

The two nutrients of principal concern from a water quality standpoint are nitrogen (in the nitrate form) and phosphorus. Both of these nutrients can accelerate the eutrophication process. Nitrate can also be hazardous to humans and animals if the concentration in drinking water exceeds 10 ppm N or 45 ppm N as

TABLE 2

Vegetable, Cotton, Soybean, and Corn Acreage in the Southeastern United States, 1972

	Vegetables	Cotton Planted	Soybeans Planted	Corn Planted
	Tons x 1000	- - - - - 1000 Acres	- - - - - -	
Alabama	126.8	600.0	780	631
Florida	1,687.9	12.5	238	365
Georgia	168.5	461.0	720	1,620
Kentucky	16.6	5.8	983	1,131
Mississippi	56.3	1,664.0	2,580	220
North Carolina	239.1	210.0	1,239	1,520
South Carolina	186.6	400.0	1,170	435
Tennessee	66.8	540.0	1,411	670
Virginia	122.7	4.8	376	706
Total, Southeast	2,671.3	3,898.1	9,497	7,298
Total, U.S.*	21,643.8	13,999.7	47,003	66,753
% of Total, Southeast	12.3	27.8	20.2	10.9

*Alaska and Hawaii are not included.

1973 Agricultural Statistics, USDA

NO_3^-, but the accelerated eutrophication of impounded surface waters is the principal concern.

Several things should be borne in mind about the behavior of nitrogen and phosphorus in soils. The only nutrient form that is readily transported by the leaching process is nitrate. Nitrogen in this form is not absorbed by the soil and thus moves readily with water. Ammonium absorbs on clays and does not leach in this form. However, all forms of nitrogen, organic and inorganic, are readily converted to nitrate when applied to productive, well-drained soils. Thus, nitrogen in the nitrate form is vulnerable to both runoff and percolation waters.

Phosphorus, on the other hand, is a highly immobile nutrient. It is not subject to leaching except in organic soils and in sandy soils having less than 5% clay. Thus, phosphorus movement is primarily associated with eroded soil particles [Barrows and Kilmer, 1963; Kilmer, 1972].

Both nitrogen and phosphorus occur in soils in the organic and inorganic form. Of the total amount of nitrogen in soils, nearly all is in organic combination. This nitrogen is released slowly as soil organic matter decomposes and is usually converted to the nitrate form. The percentage of soil phosphorus in organic combination may vary from nil to 70% of the total phosphorus present. Well-drained upland soils in the humid regions usually have 40-60% of the total phosphorus in organic form.

Rocks and Soils as Sources of Nitrogen and Phosphorus

Losses of nutrient elements have continued during rock-weathering soil formation over many millions of years. Such losses are part of a normal, continuous cycle, the nutrients disappearing in some places only to reappear in others. Indirect evidence of these losses is shown in Table 3. It is evident from this example that more phosphate was lost during soil formation from granite rock than from limestone. The concentrations of N and P in the A_1 horizons are likely due to plant cycling.

The classic work of McHargue and Peter [1921] in Kentucky, quoted by Thomas [1970], shows that phosphorus in waters is directly related to the phosphorus content of the soils and the rocks from which the soils were formed [Table 4].

In general the native nitrogen content of southeastern soils is relatively low as is seen in Table 5 [USDA Yearbook of Agriculture, 1938]. Since native soil supplies of nitrogen and phosphorus are low in most southeastern soils, transport of these nutrients in the solution phase from unfertilized soils does not appear to pose a significant pollution hazard to surface and ground waters. For example, studies in North Carolina showed that unfertilized, steeply sloping pastured watersheds lost about 4 lb/acre of nitrogen annually and less than 0.2 lb/acre of phosphorus [Kilmer et al., 1974].

Precipitation and Fallout

Precipitation is a source of plant nutrients that is frequently overlooked. Often nutrient additions from precipitation are significant from the standpoint of accelerated eutrophication. A recent report by the U.S. Geological Survey

TABLE 3

Proportions of N and P in Two Ultisols (Red-Yellow Podsolic Soils) and in Fresh Granite (Georgia) and Limestone (Virginia)

	Developed from Granite		Developed from Limestone	
Horizon	N	P	N	P
	%			
A_1	0.26	0.04	0.16	0.024
A_2	0.04	0.02	0.03	0.007
B_1	0.02	0.02	0.02	0.014
C	--	0.01	0.02	0.014
Fresh rock	--	0.17	--	0.014

Adapted from Simmonson (21).

[1974] indicates that the annual rate of nutrient deposition in northeastern basins and watersheds exceeds the rate of removal in drainage waters. This conclusion has been substantiated in other recent nutrient balance studies. Recent work in North Carolina [Kilmer, et al., 1974] has shown that annual atmospheric additions of nitrogen, phosphorus, potassium, and sulfur were often 10 times as much as drainage water removals. There seems to be little question that atmospheric additions of plant nutrients are gradually increasing due to accelerated industrialization and possibly increased use of fertilizers.

Fertilizers

Chemical fertilizers account for at least one-third of the agricultural production in the United States. The necessity for their use in supplementing the nutrients supplied by the soil has long been recognized. The use of fertilizers permits optimum crop yields, enhances crop quality, and reduces erosion hazards through the establishment and maintenance of a vigorous vegetative cover.

TABLE 4

Concentration of Phosphorus in Several Streams and Parent Rock through Which They Run

Stream	Draining	ppm P
North Elkhorn Creek (Kentucky)	High phosphate limestone	0.22
Green River (Kentucky)	Low phosphate limestone	0.07
Kentucky River (Upper)	Sandstone and shale	trace
Kentucky River (Lower)	High phosphate limestone	0.09
Ohio River, Paducah	Mixed	0.07

Adapted from Thomas, (1970).

TABLE 5

Average Soil Nitrogen Content in Two Regions of the United States

Region	Area	Average Amount of Soil Nitrogen to a Depth of 100 cm
	1×10^6 Acres	- - - - - - - - - kg - - - - - - - -
South Atlantic and Gulf States	150	1800
Southern Midwestern and Prairie States	263	7300

Adapted from Soils and Men, p.366.

TABLE 6

Plant Nutrient Consumption in the Southeastern States, 1950 and 1970

Nutrient	1950	1974 (Prelim.)
	- - - - - - - - - tons	- - - - - - - - - -
Nitrogen (N)	468,269	1,540,609
Phosphorus (P_2O_5)	759,034	952,739
Potassium (K_2O)	465,737	1,316.319
Total	1,693,040	3,809,667

Adapted from Hargett, Fertilizer Summary Data, (1974).

As a general proposition, it is difficult for an agronomist to recommend rates of fertilizers that will precisely meet crop needs because of the numerous fertilizer-soil reactions that occur. Consequently, farmers tend to over-fertilize so that nutrients are not the limiting factor in crop production. With respect to fertilizer efficiency, the crop grown in the same year that nutrients are applied will utilize about 50-60% of the applied nitrogen, 15-20% of the phosphorus, and 35-40% of the potassium. Nitrogen is of primary concern because it is the principal nutrient limiting crop yields in the U.S. and it moves readily in the nitrate form. The fate of 100 pounds of nitrogen applied to soil can be represented for the year of application as follows:

- Taken up by the crop--50 pounds
- Retained in the soil organic fraction--20 to 40 pounds
- Lost by denitrification--10 to 30 pounds
- Lost by leaching, runoff, and erosion--unknown

The amount of nitrogen taken up by the crop is well documented and, of course, there is considerable range in crop recovery. The amount retained in the soil is also quite well documented. The big holes in our understanding of nitrogen balance under field conditions lie in the areas of denitrification, runoff, and percolation. When surface-applied, nutrients are vulnerable to losses in surface runoff, particularly if a "runoff rain" occurs soon after application. Fertilizers that are surface applied are also more susceptible to losses by erosion than fertilizers mixed with the soil.

The use of plant nutrients in the Southeast has more than doubled since 1950. Nitrogen and potassium show the largest gains, while phosphorus shows the smallest [Table 6]. The figures in Table 6 probably reflect the facts that (1) nitrogen is the principal element limiting crop production, and (2) the immobility of phosphorus causes it to build up in agricultural soils.

As a general proposition, the pollution potential from fertilizers is greatest where large acreages receive high rates of fertilizers. Low fertilization rates on large acreages, or high fertilization rates on low acreages, will not have a significant overall effect on water quality, except in a localized sense.

The average farm size in the southeastern states is 169 acres, ranging from 103 acres in South Carolina to 426 acres in Florida [Agricultural Statistics, 1973]. The average farm size in the Corn Belt states is 197 acres. The average farm size is not grossly different in the two areas and the application rates of nitrogen and phosphorus are about the same. The Midwest, however, has about three times as much agricultural cropland as does the Southeast. The Southeast would therefore appear to run a lower risk of pollution from fertilizers than some other areas of the U.S.

Animal and Poultry Wastes

Prior to the beginning of World War II, animal manures provided a major source of supplemental nitrogen. Presently, it is estimated that about 1 million tons of nitrogen are applied to agricultural lands in the form of manure. These applications benefit the soil by improving fertility and structure. Society is benefited since application to soils is a partial solution to the animal waste disposal problem.

The utilization of animal wastes as a nutrient source poses problems. These materials are bulky, low-grade fertilizers of variable composition. The total plant nutrient content is only 10 to 20% of that found in most commercial fertilizers. Concentrations of nutrients, soluble salts, trace elements, and water content vary tremendously and are seldom known. Optimum rates of application are thus difficult to predict. The pollution potential from using manure with poor management can be substantially higher than from use of commercial fertilizers. Nearly all manure is spread on the soil surface and can contain large amounts of soluble carbon, nitrogen, and phosphorus compounds. These soluble constituents are easily lost if runoff occurs before incorporation into the soil. In this respect, manure applied to frozen ground is particularly vulnerable.

Volatilization of ammonia from concentrated animal feedlots into the atmosphere, with subsequent absorption in adjacent surface waters, is also a problem [Hutchison and Viets, 1969; Luels, et al., 1973]. The extent of this problem is unknown in the Southeast and should be studied.

The principal concentrated sources of manure are confined beef and dairy cattle, swine, and poultry operations. Strictly speaking these are "point" sources, but, once the wastes are spread on land, they may be considered "non-point" sources. The pollution hazard from non-confined stock is unknown, but probably is not great since the concentration of grass-fed animals seldom exceeds one-third to one-half animal units per acre.

TABLE 7

Cattle and Poultry Production in the Southeastern United States, 1972

	All Cattle Plus Calves	All Cattle Plus Calves on Feed	Swine	Chickens (Broilers)
	- - - - - - - 1,000's	- - - - - - - - - -		millions
Alabama	722	37	1,000	399.2
Florida	608	58	310	56.6
Georgia	730	65	1,982	442.9
Kentucky	1,122	38	1,352	7.2
Mississippi	698	14	487	256.2
North Carolina	330	45	1,780	301.7
South Carolina	210	26	710	28.2
Tennessee	789	14	1,155	54.9
Virginia	575	39	644	77.2
Total, Southeast	5,784	336	9,420	1,624.1
Total U.S.*	51,462	14,485	61,502	3,074.7
% of Total, Southeast	11.2	2.3	15.3	52.8

*Alaska and Hawaii are not included.

From Agricultural Statistics, 1973.

Cattle, swine, and chicken production in the southeast are shown in Table 7. Taken as a whole, the southeastern states have about 11% of the nation's milk. A little over two percent of the cattle are on confined feeding. Disposal of wastes from swine and chicken production appears to be the principal problem facing southeastern states at this point in time.

Pesticides

Pesticide use has increased sharply over the past 15 years from around 600 million pounds in 1960 to well over a billion pounds at present. The behavior of pesticides from an environmental standpoint is an exceedingly complex matter, and I must confess at the outset that I am rather ill-equipped to discuss the subject, except in a very general way. For an excellent basic review of the subject, the reader is referred to Pionke and Chester [1973].

Loss mechanisms of pesticides are similar to those affecting nitrate, in that movement out of the soil profile can be through leaching, surface runoff, soil erosion, and volatilization.

The potential leaching characteristic of a given pesticide depends primarily upon how strongly it is absorbed on soil colloids and upon hydrologic factors. Weakly absorbed pesticides are, of course, most vulnerable to leaching; those that are strongly held are most susceptible to erosion losses. As a general proposition, the acidic herbicides as a group (DNPB, pyriclor, norea, etc.) are the most mobile; chlorinated hydrocarbon insecticides (aldrin, dieldrin, endrin, heptachlor, etc.) are the least mobile. The persistence in soils of selected pesticides is shown in Table 8.

Numerous investigations have been carried out involving losses of agricultural pesticides from soil systems via the various loss mechanisms. The main conclusion to be drawn from these investigations seems to be that pesticide concentrations in surface runoff are generally very low and such losses do not exceed 5% of the application during a given crop-year. In spite of this, many pesticides at exceedingly low concentrations are extremely toxic to fish and other aquatic life. However, unless a heavy rainfall occurs immediately after application, there appears to be a minimum pollution hazard from pesticides if properly used. It is just the local accidents with heavy fish kills that attract public attention. Scientists are learning how to prevent these catastrophes and, in time, it is hoped that such occurrences will be rare.

I can come to no astounding conclusion regarding the pesticide pollution hazard as it relates to the Southeast. Considering the amounts used and the acreage involved, the problem seems to be slight.

Sediment

It is fairly well-agreed among soil conservationists that sediment is not only the greatest single water pollutant in the United States but also the largest carrier of agricultural chemicals. Sediment is estimated to exceed the sewage load in streams by 500-700 times [Glymph and Carlson, 1966]. Sources of sediment are farmland, construction sites, stream banks, and road banks. It is estimated that 4 billion tons of sediment are washed annually into United

TABLE 8

Approximate Time Required for the Disappearance of 50% and 90% of Selected Pesticides Applied to Soils

	50%	90%
Chemical	- - average number of days - - -	
s-Triazines		
Atrazine	130 ± 40	294
Simazine	105 ± 34	360
Chlorinated Hydrocarbons		
DDT (aerobic)	3837 ± 2231	6281
Endrin	4342 ± 2049	--
Chlorodane	2951	--
Phosphates		
Methyl parathion	4.4	26
Malathion	0.56 ± 0.41	1.85 ± 1.27
Carboxylic acids		
2-4-D	17 ± 8	26 ± 10
Dalapon	14.6 ± 5.9	27 ± 16

Adapted from Hamaker, 1972.

States waterways and reservoirs. Fifty percent of the sediment is attributed to agricultural endeavors [Robinson, 1971]. A monumental amount of information on sediment losses has been accumulated by federal and state scientists since about 1920, culminating in Wischmeier and Smith's [1960] formulation of a Universal Soil Loss Equation which was a major breakthrough in soil conservation.

Soil conservationists recognize that there is a residual soil loss from farmland that is not presently reducible. This is called a "tolerable soil loss." For typical midwestern soils, such as those found in central Illinois, it may be in the range of 6 to 10 tons/ha/year [Stall, 1972].

A highly useful map of the United States has been prepared showing the concentration of sediment in mainland rivers of the United States [Rainwater, 1972]. As a gross generalization, the highest sediment yields in the U.S. occur in the sand hills of western Mississippi and amount to 15 tons/ha/year. Table 9 shows the distribution of southeastern cropland on which water erosion is a problem. If the annual tolerable soil loss is set at 3 tons/acre, an average of 62% of southeastern cropland has a mild erosion problem, 22% a moderate problem; and 11% a severe problem. Scientists have known for a long time that eroded material frequently differs in composition from the original soil. The finer clay and organic matter particles are the most absorptive fraction of the soil. They are also the lightest and therefore tend to erode first, carrying sorbed chemicals. Thus, the eroded material generally has a higher concentration of nutrients and pesticides than the soil from which it originated.

Suspended sediment tends to act as a cycling mechanism in streams and lakes, particularly where phosphorus is concerned. When soluble phosphorus concentrations exceed a certain level in water--thought to be around 0.01 ppm--phosphorus is absorbed by sediment. When soluble phosphorus levels are reduced, sediment may release phosphorus to water.

There appears to be little question that vast amounts of chemicals are added to southeastern surface waters each year via the soil erosion process. Erosion control is the answer to this problem.

SUMMARY AND PERSPECTIVE

The southeastern region of the United States possesses topographic, climatic, and hydrologic characteristics that favor the transport of non-point, agriculturally related pollutants to surface and ground waters. On the other hand, the southeastern states utilize less than 20% of the total land area for agricultural production; the farms are relatively small, averaging 169 acres, and the long growing season, particularly in the lower South, permits a vegetative cover on the land during the major portion of the year.

Available evidence indicates that nutrient loading of streams is not yet a serious problem except in localized situations. For one thing, the unfertilized soils of the southeastern states are generally low in nitrogen and phosphorus. Concentrations of these elements in major streams appear to be holding quite steady or, at the most, increasing at a slow rate. The marine beaches along the South Atlantic Coast and Gulf of Mexico are among the cleanest in the nation [Tihansky, 1974].

The spiraling cost of fertilizer production probably will check the present tendency to use too much fertilizer. Future farmers may find that attempts to attain maximum crop yields are increasingly unprofitable.

Relatively few beef cattle are finished on grain, and disposal of wastes from these operations appears to present few problems. Present or potential pollution hazards appear to be highest in dairying operations and in pork and broiler production. The southeastern states account for over half of the total U.S. broiler production and disposal of wastes from these enterprises presents special problems.

TABLE 9

Distribution of Southeastern U.S. Cropland on Which Water Erosion Is a Problem

State	Total Land Area*	Total Cropland*	Percentage of Cropland on Which Average Annual Soil Loss Exceeds:**		
	- - (1,000,000	Acres) - -	3 Tons/A	8 Tons/A	15 Tons/A
Alabama	32.6	5.1	43	25	21
Florida	34.7	3.8	77	15	7
Georgia	37.3	6.5	60	22	13
Kentucky	25.5	6.6	60	22	17
Mississippi	30.3	7.3	83	6	4
North Carolina	31.3	6.5	55	17	16
South Carolina	19.3	3.9	52	42	5
Tennessee	26.4	7.0	70	15	10
Virginia	25.5	4.0	60	30	9
Total	262.9	50.7	62	22	11

*From USDA Statistical Bulletin No. 41, National Inventory of Soil and Water Conservation Needs, 1967.

**Based on application of the universal soil loss equation to data obtained from a nationwide sampling for the 1967 USDA inventory.

The diagnosis of pesticide use and potential hazards is a very complex problem. Each pesticide and condition of use must be studied as its own special case. The consensus of workers in the field seems to indicate that the proper use of pesticides involves a low risk as far as pollution hazards are concerned. However, there appears to be sufficient evidence to warrant an expansion of present extension educational activities relating to proper pesticide use under farming conditions.

Soil erosion is said to be the outstanding pollution problem not only in the Southeast but in the entire country. The Southeast continues to have its share of erosion problems. Not only is sediment, per se, a viable pollutant, but the auxilary components--such as nitrogen, phosphorus, and pesticides--contribute their special problems. However, suspended sediment in streams is not all on the red-ink side of the ledger. Sediment can act as a "sink" for phosphorus as well as pesticides and certain of their degradation products. Sediment may also alter the light relationships in streams and lakes, thus slowing seed germination and growth of certain aquatic plants.

What of the future? The present situation reflects a rapidly rising population and a cropland acreage that has remained fairly constant and has even decreased in recent years, as shown in Figure 1. The demand for agricultural production is certain to keep increasing. Plant protein, mainly from soybeans, will be increasingly used as a substitute for animal protein. The southeastern states in the foreseeable future probably will expand soybean acreage greatly at the expense of other crops. Soybeans require only low levels of fertilization and, being a legume, fix nitrogen from the atmosphere. Unless properly handled, soybeans and other legumes can accelerate the nitrogen loading of surface and ground waters. Soybeans also increase the soil's vulnerablility to erosion.

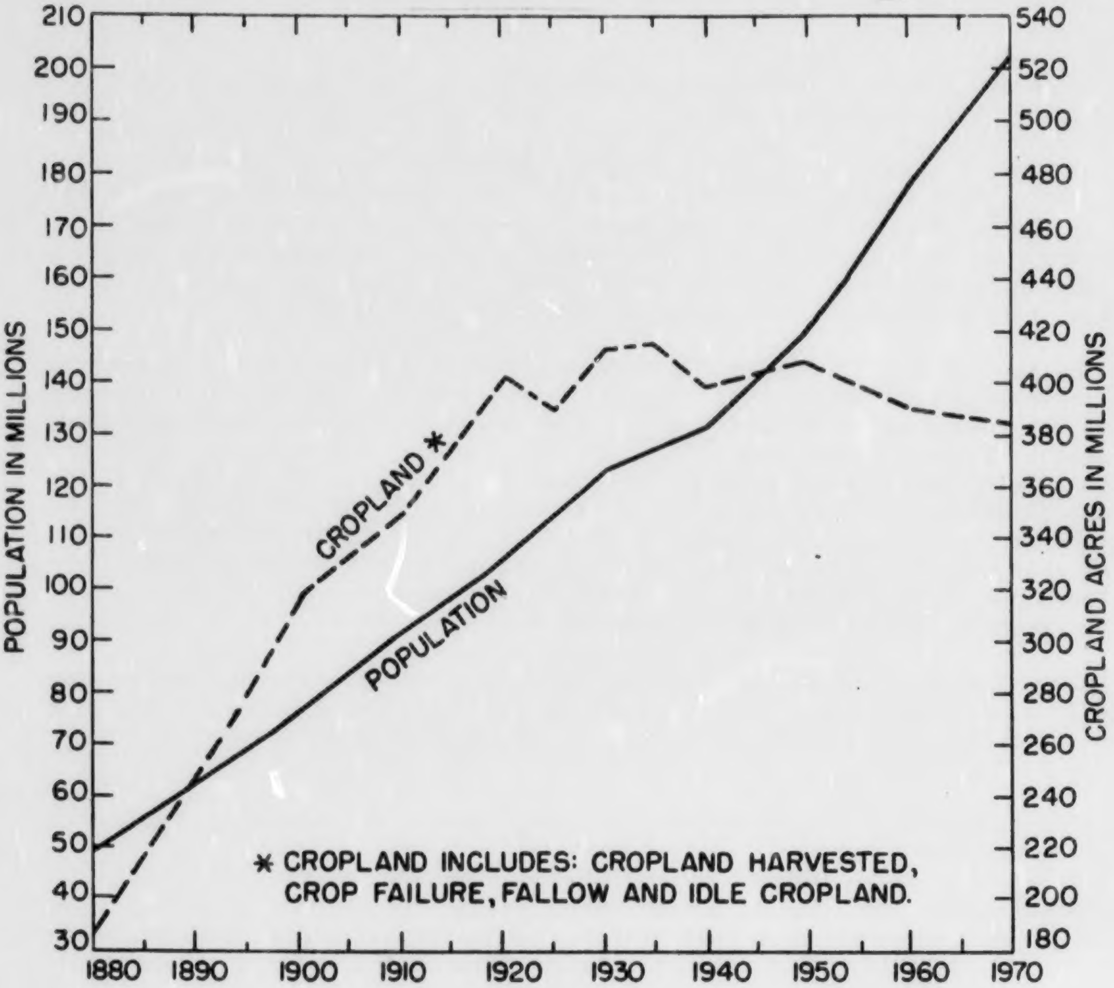
If weed and other problems can be eliminated, minimum tillage will replace our present tillage practices. There is strong evidence that minimum tillage also enhances nitrogen transport from soil because of altered moisture conditions in the soil profile.

The Southeast will continue to produce feeder cattle, but no major increase in grain-fed cattle is seen.

The major future expansion of needed agricultural production will probably take place in the midwestern Corn Belt states. Soils in that region tend to be more fertile and more responsive to fertilization than do the majority of the soils in the Southeast.

FIGURE 1

Changes in the Cropland-Population Ratio
of the United States, 1880-1970



(USDA HANDBOOK 153 AND AGRICULTURAL STATISTICS, 1973)

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SOUTHEASTERN FORESTS AND THE PROBLEM OF NON-POINT SOURCES OF WATER POLLUTION

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ABSTRACT

The 1972 Amendments to the Federal Water Pollution Control Act require that non-point source pollution from forestry activities be controlled. Erosion is the most serious pollutant originating from forestry activities. Activities which destroy the forest floor and reduce the infiltration rate of soil, such as mechanically preparing sites for tree planting, roads, skid trails, and logging decks, cause most accelerated erosion. Elevation of water temperature, changing the chemical composition of water by cutting or fertilizing stands, and introduction of pesticides and herbicides into streams are other common forms of pollution. Pollution from forestry operations can be minimized by utilizing existing information. However, there is need for additional quantification of pollution levels associated with alternative forestry practices, and for screening of known or development of new techniques for minimizing non-point source pollution.

INTRODUCTION

For years, foresters felt smug about forestry practices and water quality. They praised the forest for its role in regulating streamflow, binding soil in place, rehabilitating depleted soils, and serving as the source for high-quality water. They pointed their finger at downstream neighbors and said that agriculture, highways, construction, municipalities, and industries were the defilers of our priceless heritage of clean water.

But foresters knew that woods roads, logging, and other practices could accelerate erosion and thereby degrade water quality. More recently, large changes in chemical composition of water were noted after a Northeastern hardwood stand was clearcut and kept bare with herbicides for three years. These changes also worried foresters. The bubble finally burst with passage of the 1972 Federal Water Pollution Control Act Amendments. Now foresters--along with farmers, highway departments, municipalities, and industries--are to be held accountable for "non-point source pollution," and the shock still has not worn off.

The Environmental Protection Agency now is attempting to assess the magnitude of the problem of non-point sources of pollution, and foresters seriously are asking what kinds of and how much pollution are created by forestry operations.

The 1972 Act made these questions paramount in importance to the forest industry. The ultimate goal of the legislation was "no discharge" by 1985. However, EPA recognizes that the "no discharge" goal cannot be implemented under the existing Act; furthermore, it foresees that universal achievement of this goal may not be either feasible or environmentally desirable. A more practical goal is a level of water quality which will support fishing and swimming by 1983 [U.S. Environmental Protection Agency, 1974]. But, even here, the "where attainable" stipulation recognizes that naturally occurring conditions, or uncontrollable pollution from non-point sources, could result in a failure to meet this goal everywhere.

Where, then, does this Act leave the forestry profession with respect to non-point source pollution? The law requires that the "best practical technology" for combating non-point source pollution be applied in the period up to 1977; thereafter, EPA will exert greater emphasis on land-use regulation and on the establishment of legal requirements for forestry and other operations which produce pollution from diffuse and intermittent sources. Thus, assuming that the best practical technology is being applied today (and this assumption is questionable), the industry has a breathing spell in which to develop and implement the "best available" technology required by 1983.

What constitutes non-point source pollution to water courses? Better yet, what is water pollution? One EPA official defined pollution as "a resource out of place." This definition is too general to be of much value and yet is still not all-inclusive. A better definition might be that water pollution is any introduction of foreign material into water or other impingement upon water which produces undesirable changes in the physical, biological, or chemical characteristics of that water. For the purposes of discussion, I wish to consider non-point source pollution with respect to this definition.

SOIL AS A POLLUTANT

Erosion is the process by which soil and minerals are detached and transported by water, wind, and gravity. Subsurface erosion, the eluviation of the earth's mantle, is a natural weathering process, and dissolved minerals comprise one-third to one-half of the total load of Southeastern streams. Surface erosion, the detachment and movement of particles over the surface of the ground, is strongly influenced by management practices, climate, soils and topography. Generally, less than 300 pounds of suspended load and bedload per acre are lost from forested lands of the Southeast. This load does not come uniformly from each acre of forest land; rather, it originates primarily from natural geologic cutting of banks and channels. Now, let us examine the erosion process and how the forest and its management affects erosion and non-point source pollution.

CLIMATE

Erosion occurs only because energy for detachment and transport of soil particles exceeds the cohesive and adhesive forces which bind soil in place. In the East, the energy for erosion comes principally from rainfall and flowing water.

The greatest source of energy is from falling raindrops. Although discussed many times, the tremendous erosive force of raindrops is rarely appreciated and is due primarily to their velocity. Raindrops reach a maximum diameter of only about 5 mm, but they strike exposed soil like tiny bombs. They "explode" digging craters and blasting soil into surrounding areas. An inch of moderately intense rainfall delivers over 1,000 times more potential energy for erosion than an equivalent volume of water flowing uniformly over the soil surface.

Rainfall throughout most of the South ranges from 45 to 55 inches annually, decreasing westward through Texas and increasing in the lower Blue Ridge Province to as much as 80 inches annually [U.S. Department of Commerce, 1968]. Rain falls on 100 to 120 separate days throughout most of the South.

Rainfall intensities are greatest along the Gulf and Atlantic Coasts and generally decline to the interior [U.S. Department of Commerce, 1961]. Thirty-minute storms of about 1.5 to 2.5 inches occur about 1 year in 10 (the smaller amount being at the higher latitudes). Both drop size and velocity increase with rainfall intensity [Laws, 1940, 1941; Laws and Parsons, 1943]. Velocity is about 13 feet per second for drops 1 mm in diameter and increases to a maximum of 26 to 28 feet per second for 4 mm drops [Bennett et al., 1951].

Orographic rains of the Appalachians normally have small drop size. Larger drops are associated with warm rather than cold frontal rains [Cataneo and Stout, 1968; Mueller et al., 1967; Sims et al., 1964]. Because cold frontal storms predominate in the winter, winter rains have lower erosive energy than convection storms, which generally occur during summer and have the highest intensities and large drop sizes. In the South, 30 to 50 percent of the storms are the convection type. Calculations based on Laws and Parsons' data [1943] for raindrop size and intensity of low-intensity rains (0.05 inch per hour) and data on velocity from Bennett et al. [1951] indicate that the energy of high-intensity raindrops may be 40 times greater than that for low-intensity rainfall.

The Agricultural Research Service has developed a map showing the erosion index for average rainfall in the United States east of the Rocky Mountains. The erosion index is the product of the total kinetic energy of the storm times its maximum intensity. In the Southeast, the index ranges from about 200 to 500, indicating that, if other factors are held constant, erosion in the Southeast might vary by a factor of 2.5 because of rainfall differences.

Rain which does not infiltrate the soil moves overland and possesses kinetic energy for detachment and transport of soils. Total energy is influenced mostly by velocity. The weight and size of material which can be carried by runoff varies with the 5th and 6th power of velocity. Thus, in both detachment and transport of soil, velocity of flowing water is a key factor. Velocity of overland flow, if it occurs under forest stands, is slow. But since overland flow is concentrated in rills and gullies, the velocity of flow, depth of flow, or both are increased, the energy is concentrated, and erosion losses increase sharply [Ellison, 1945].

Alternate freezing and thawing of soils also exert strong forces for the detachment of soil particles. Concrete frost usually does not occur under forest stands in the South, but it may occur in bare soils at higher elevations and latitudes. When concrete frost occurs, it reduces infiltration and increases overland flow [Pierce et al., 1958; Post and Dreibelbis, 1942; Sartz, 1957; Trimble et al., 1958]. Frost heaving (stalactite frost) of exposed soil is a greater problem in the South. During freezing, water is drawn by capillary action from greater depths, and ice crystals develop vertically. Peds, rocks, and soil particles are lifted during freezing. After crystals melt, the displaced particles are deposited in a loose layer ready for further disintegration and transportation by water.

High summer temperatures desiccate bare soil, and a layer of loose, structureless soil similar to that caused by frost heaving is produced. Thus, temperature extremes set the stage for high erosion losses from bare soils during rainstorms.

While we cannot control the climate of Southern forests, we can control water erosion by following two rules:

1. Protect the soil from raindrop impact, frost action, and high temperatures, and

2. Minimize the volume and velocity of water which flows over the soil surface.

SOIL ERODIBILITY

Soils vary in erodibility, depending on their properties. Middleton [1930] found that the most significant properties related to erodibility were (1) the dispersion ratio, and (2) this ratio divided by the ratio of colloid to moisture equivalent. He suggested that organic matter, silica/sesquioxide ratio, and total exchangeable bases also influenced erosion. Later, Anderson [1954] successfully related Middleton's dispersion ratio, erosion ratio, and watershed cover to turbidity on 14 California watersheds. He also found that colloids, moisture equivalent, and silt + clay were related to turbidity. Erosion also varies with moisture content, infiltration rate, and degree of aggregation [Baver, 1956]. Smith and Wischmeier [1962] concluded that soil effects were interrelated with past cropping, erosion, and topography and were automatically confounded with rainfall characteristics. After extensive efforts to characterize soil erodibility, they finally used an experimentally derived erodibility factor, K, in their erosion prediction method. This was an empirically derived ratio of soil loss per unit area from plots with specific characteristics to the total kinetic energy of the storm multiplied by the maximum 30-minute rainfall intensity. These plots were 72.6 feet long, uniformly sloping at 9 percent, in continuous fallow, and tilled up and down slope. The value of this ratio varied from 0.10 to 0.39 for most Southern soils for which K was determined. Later, K values were estimated for all soils on the basis of surface texture and permeability.

TOPOGRAPHY

Percentage and length of slope are the most significant topographic variables affecting erosion. Smith and Wischmeier [1957] combined data from four studies and found that soil loss (A) from agricultural soils fitted the relationship:

$$A = \frac{0.43 + 0.30S + 0.43S^2}{6.613}$$

where S is percentage of slope. Neal [1938], using artificial rainfall, found that soil loss in pounds from 1/1,000-acre plots was a function of slope and rainfall intensity. Zingg [1940] reported that erosion is an exponential function of both slope and length. Wischmeier and Smith [1965] used both slope and length as a multiplier in their universal equation for soil erosion of croplands.

MANAGEMENT EFFECTS

Wischmeier and Smith's [1965] equation for predicting soil loss from croplands provides insight into the relative importance of factors influencing erosion. This equation is:

$$A = RKLSCP$$

where A is soil loss in tons per acre, R is the rainfall factor, K is a soil erodibility factor, LS is the factor for length and percentage of slope, C is the crop management factor, and P is the erosion control factor. The estimated soil loss is sheet erosion from slopes, not necessarily that quantity of soil which reaches stream channels.

The most important single factor in determining soil loss is the cropping practice (C) applied to the land. Man's choice of cropping practices can produce up to a 250-fold difference in soil loss. Once the cropping factor has been selected, erosion losses can be reduced up to tenfold by conservation practices (P). The rainfall energy factor (R) can account for a threefold difference in erosion; the LS factor can increase or reduce loss about sixfold; and soil erodibility can account for a fourfold difference in soil loss. The conclusion is inescapable: Man's actions are more important in determining soil loss than all other factors combined.

HOW FORESTS AFFECT THE EROSION PROCESS

The major effect of forests on erosion is to reduce slightly the volume of rainfall reaching mineral soil and to change greatly the energy available for detachment and transport of soil particles. From 8 to 18 percent of total rainfall is intercepted by vegetation, and 2 to 4 inches of annual rainfall is intercepted by the forest floor and evaporates [Helvey, 1964, 1972; Helvey and Patric, 1965]; but the organic layers intercept only up to 0.05 inch of each rainstorm. More important, water temporarily stored in the forest floor drains slowly, thus increasing the likelihood of infiltration.

Annual litterfall is over 1 to 2 tons per acre for fully stocked hardwood stands [Blow, 1955; Bray and Gorham, 1964; Chandler, 1941; Cromack, 1973; Kovner, 1955; Metz, 1954; Sims, 1932]. Decomposition rates vary with species, but decomposition requires more than a year [Cromack, 1973]; hence, there is a buildup of 3 to 14 tons of organic layers per acre, greater weights being associated with pine forests [Metz, 1954; Ursic, 1963].

The forest floor also offers resistance to overland flow and reduces the velocity of any water which may fail to infiltrate. Because velocity is reduced, the kinetic energy, competence, and carrying power of runoff are all reduced. Stated simply, these reductions result in low rates of erosion. Wischmeier [in press] attempted to modify the Universal Equation for Soil Loss in order to account for the effect of vegetative cover and the forest floor on erosion. He assumed that raindrop energy would be dissipated because of interception, and that stem drip would not regain terminal velocity. However, foliage and branches serve as collectors of rainfall, and drop size of throughfall may equal or exceed that of rain. Drip height may be sufficient to allow terminal velocity to be reached. Because drop energy is a function of both drop size and falling distance, the erosive energy may be increased or decreased by the forest cover. Chapman [1948] showed that, for precipitation rates of less than 2 inches per hour, the kinetic energy of raindrops beneath a red pine canopy actually exceeded that in the open.

Decomposed litter is incorporated into surface layers of soil by earthworms and other soil fauna. Surface soils under forests characteristically differ from old fields by having lower volume weight, greater organic content, greater percentage of large pore space, greater detention storage capacity, and higher infiltration rates [Alderfer and Merkle, 1941; Hoover, 1944; Sartz, 1961; Trimble et al., 1951]. Infiltration rates are typically 20 to 100 inches per hour in upland forest soils. Because rainfall rarely exceeds 8 inches per hour anywhere in the East, little or no water runs off the surface, and little erosion takes place.

Johnson and Swank [1973] measured soil losses of 29 and 72 pounds per acre for mature Appalachian hardwoods and white pine. The material recovered was primarily bedload of perennially flowing streams, and much of the suspended load went unmeasured. Because bedload movement is a channel phenomenon, prorating this movement on a per-acre basis creates the illusion that each acre is losing soil when, in fact, most of the soil comes from the channel and bank scour.

McClurkin [1970] found that, even on eroded soils reclaimed to pine, physical properties of soils improved and, after 15 years, infiltration rates had increased by an average factor of 2.5, or to about 5 inches per hour. Erosion losses from second-growth hardwoods in the Piedmont averaged 3.2 pounds per acre for 7 years as compared with 21.8 and 28.7 tons from 4-year rotations of corn and cotton [Copley et al., 1946]. Ursic [1965] reported that, as a result of channel cutting in Mississippi, soil losses over a 3-year period averaged 200 pounds per acre from three watersheds on depleted hardwood sites, 40 pounds per acre from three watersheds consisting of pine plantations established for erosion control, and 40 pounds per acre from three watersheds supporting forests of mature pines and hardwoods. Ursic and Duffy [1972] found that sediment production averaged 13 pounds per acre-inch of runoff from 30 year-old loblolly pine growing on seven small, severely eroded catchments. Rogerson [1971] determined that annual soil loss over a 9-year period averaged 13.8 pounds from three 1.4- to 1.6-acre pine-covered watersheds in Arkansas. These examples illustrate the low annual rate of erosion which takes place from stable forests of the South.

PRACTICES WHICH INCREASE SOIL LOSSES

Increases in erosion have been associated with cutting the forest, but if the forest floor is not removed or grossly disturbed by cutting the overstory, infiltration rates remain high and overland flow is virtually absent [Hoover, 1944; Hornbeck et al., 1970; Johnson and Kovner, 1954; Reinhart, 1964]. Even after clearcutting, litter and debris provide protection for several years, depending on the forest floor present, the litter decomposition rate, and the litter production rate. Herbaceous grass, seedling, and sprout cover develop rapidly after cutting. In three to four years, the stands of regrowing sprouts provide nearly a 100-percent canopy cover and buildup of forest floor.

When the forest floor is removed by the building of roads, the skidding of logs, or wildfire, raindrop energy is dissipated on mineral soil. Infiltration rates are reduced by compaction and puddling of the soil. When infiltration rate falls below the rainfall rate, overland flow and sheet and rill erosion occur.

Packer [1967] reviewed the literature and concluded that erosion from forest roads was the major source of suspended solids in streams of forested areas. Undeniably, erosion from poorly designed, constructed, and maintained roads is an important contributor to water pollution [Anderson, 1954; Douglass and Swank, in press; Hoover, 1945; Lieberman and Hoover, 1948a, 1948b; Megahan, 1972; Reinhart et al., 1963]. Erosion losses reported in the literature are sometimes the "horrible examples" which flag the importance of roads as a potential source of pollution. There is, however, little quantitative data on weight of soil lost from different types and classes of roads and skid trails with which to assess the magnitude of the overall problem as it exists today. But we know that when causes of erosion are adequately considered in the design, construction, stabilization, and maintenance of roads, erosion from roads can be greatly reduced [Hewlett and Douglass, 1968; Kochenderfer, 1970]. The extent of control is largely a matter of subjective judgment because quantitative data from the "good" examples are generally lacking.

Disturbances during logging can remove the protective organic layer and compact the soils. The disturbance varies with the silvicultural system, the layout and supervision during logging, and the type of equipment used [Campbell et al., 1973; Dickerson, 1968; Hatchell et al., 1970; Moehring and Rawls, 1970]. Measurements that would enable us to determine the relative importance of these factors or to quantify soil losses have not yet been made, although Dickerson [unpublished report] found that neither overland flow nor sediment production from skid trails on erosive Coastal Plain slopes posed a serious threat to water quality when viewed from a catchment basis.

Results from studies of the effects of fire on erosion are conflicting. Repeated use of prescribed burning was not reported to have caused significant changes in hydrologic properties of Coastal Plain soils in South Carolina [Metz et al., 1961] or Piedmont soils in Georgia [Brender and Cooper, 1968]. Nor was it reported to increase soil movement in established gullies of the South Carolina Piedmont [Cushwa et al., 1971]. However, others report that burning has reduced infiltration rates, increased sediment yields and, in one case, caused reactivation of a gully [Auten, 1934; Copley et al., 1946; Paulsell, 1957]. Prescribed burning of old fields in Mississippi increased sediment to 1,600 and 5,800 pounds per acre, the higher value being associated with reactivation of an old gully [Ursic, 1969]. By the third year, soil losses from burned watersheds dropped to less than 50 pounds per acre. Prescribed burning and deadening of depleted upland hardwoods increased soil loss from two catchments by 49 and 119 percent the first year, but the maximum increase in soil loss was less than 500 pounds per acre [Ursic, 1970].

The bulk of the evidence indicates that occasional burns for vegetative control can increase soil loss (normally less than 300 pounds per acre) up to threefold. Grass, herbs, and sprouts are often sufficient to provide a ground cover within a few months, and erosion decreases to near normal within a few years. However, annual (as opposed to periodic) prescribed burning deprives the soil of a litter layer, and a progressive reduction in infiltration rates and increases in surface runoff and erosion can be expected. Erosion from burned plots in a second-growth forest of shortleaf pines and hardwoods in the Piedmont increased progressively from about 2 pounds per acre to 7.8 tons per acre over a 7-year period [Copley et al., 1946].

Quantitative data on soil loss from various silvicultural practices have not been reported for Southern conditions. Reinhart et al. [1963] did find large differences between maximum turbidities after selection cuttings where roads were carefully planned and those after commercial clearcutting and diameter-limit cuttings where roads and logging methods were left to the logger's discretion.

Quantitative data are also unavailable for soil losses after site preparations including tree crushing, bedding on slopes, root raking, or combinations of these. Using modifications of the modified Musgrave procedure, Dissmeyer [1973] estimated that 80 percent of all forestry-related production of sediment in the Santee River Basin was due to mechanical site preparation. Even erosive losses from roads can be dwarfed by comparison with the losses from such preparation.

OTHER FORMS OF POLLUTION

Selective or complete cutting of the forest increases total flow and changes the timing of flow. Models are available which predict the magnitude and duration of increases in annual flow [Douglass and Swank, 1972 (in press)]. Clearcutting leads to slight increases in peak discharge during storm periods [Hewlett and Helvey, 1970]. Partial cuttings that result in increased soil moisture and streamflow also tend to increase flow during storm periods. Higher flows are usually accompanied by higher velocities, indicating that both carrying power and competence of streams are greater for a period of time after forest cuttings. An increase in channel and bank erosion is to be expected, but an increase in erosion due solely to increased flow from cutting stands has not been demonstrated. Because increases in discharge due only to cutting of timber are small, increases in channel and bank cutting from such discharge should not be great. Practices which cause or increase surface runoff, such as fire, roads, and skid trails, tend to increase peak discharge and presumably channel and bank erosion.

Cutting all vegetation along streams can increase stream temperature until a shade cover regrows [Likens et al., 1970; Lynch, 1969; Reinhart et al., 1973; Swift and Messer, 1971]. Increases in water temperature can be critical in fish management. However, thermal pollution can be prevented by leaving shade strips along streams. Water that is heated in forest openings cools rather quickly after flowing through uncut or shaded reaches of streams [Swift and Baker, 1973]. Thus, even though temperatures can be increased by opening up the stream channel, techniques are available to minimize or prevent this form of pollution.

Less is known about chemical changes in ionic concentration of streams after forest cuttings. Increased export of nutrients after clearcutting and suppression of regrowth with herbicides for 3 years was reported in New Hampshire [Likens et al., 1970]. Nitrate losses increased sharply and, at times, exceeded standards for drinking water. Accelerated loss was noted for Ca, Mg, K, Na, Al, NO_3, SiO_2, and Cl, but concentrations of SO_4-S, NH_4, and HCO_3 decreased. In western North Carolina, detailed studies of the nutrient balance in an undisturbed forest, a grass-to-hardwood forest, a young plantation of white pine, a regrowing coppice forest, and in other watersheds subjected to various cutting practices have not shown appreciable changes in the ion budget except for nitrates [Douglass and Swank, in press; Johnson and Swank, 1973]. Average annual concentrations of NO_3-N were increased by factors of 4 to 150, but at no time did NO_3-N, or any other ion measured, approach a concentration hazardous to human health. In West Virginia, clearcutting a hardwood forest and then allowing immediate natural regeneration had a negligible effect on most cations and anions measured. Only NO_3-N and phosphate concentrations showed slight increases during storm periods, and sulfate showed reductions [Aubertin and Patric, 1974]. Whether small changes in nutrient concentration are harmful or beneficial to aquatic communities is not known at this time. The changes so far observed in water chemistry from timber harvest in the Southeast are not cause for alarm.

Herbicides and pesticides are also potential sources of pollution. Douglass et al. [1969] observed that a narrow unsprayed strip was sufficient to prevent pollution of streams from atrazine and 2,4,5-T. Similarly, Norris [1971] concluded that common chemicals for brush control can be used with minimum hazard. Grzenda et al. [1964] found that method and location of application determined whether or not DDT residues appeared in streamflow after spraying. As with thermal pollution, methods of minimizing the hazards from these chemicals are known.

Fertilizers are increasingly used in forestry. These chemicals can reach streams by attaching themselves to soil particles or in overland flow [Hilmon and Douglass, 1968], from direct application to streams [Aubertin et al., 1973], and as leachate from subsurface flow [Hornbeck and Pierce, 1973]. However, the exchange capacity of soil serves as an effective filter or trap for many chemicals.

In one of the few documented studies of the effects of forest fertilization on stream chemistry, Aubertin et al. [1973] observed that urea fertilization produced immediate increases in NO_3-N and ammonium-N concentrations. The first year after fertilization, NO_3-N increased tenfold and exceeded, at times, the standards for drinking water, and ammonium-N decreased fiftyfold. Calculations indicate that 17.8 percent of the applied N appeared in streamflow. Ca, Mg, K, Cu, and Zn concentrations increased, and Fe, P, and Mm decreased, although these changes cannot be attributed wholly to urea fertilization.

In addition to the more common pollutants, salts and petroleum products, leachates from sawdust piles, human wastes, application of treated sewage to forest lands, and leachates from land fills are also possible sources of pollution from forest lands.

CONCLUSION

Erosion is the most important source of pollution during forestry operations. Normal erosion from fully stocked forests is characteristically less than 300 pounds per acre and originates mainly from normal geologic cutting of channels and banks. Management of the forest will always tend to increase erosion. Those practices which destroy the forest floor, compact the soil, and reduce the infiltration capacity of soils are most damaging because they expose soil to raindrop impact and temperature extremes and tend to increase overland flow. Accelerated erosion occurs on sites mechanically prepared for planting, roads, skid trails, and logging decks. Besides erosion, heating of water and its contamination by pesticides, herbicides, and fertilizer are common potential pollutants. However, with the information already at hand, erosion and other forms of non-point source pollution can be minimized by careful planning and supervision of forest operations. Meanwhile, there is a pressing need for quantification of pollution levels from alternative practices. Above all, we must screen known techniques or develop new techniques for minimizing non-point source pollution from forestry activities.

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This paper reports research involving pesticides. It does not contain recommendations for their use, nor does it imply that the uses discussed here have been registered. All uses of pesticides must be registered by appropriate State and/or Federal agencies before they can be recommended.

CAUTION: Pesticides can be injurious to humans, domestic animals, desirable plants, and fish or other wildlife--if they are not handled or applied properly. Use all pesticides selectively and carefully. Follow recommended practices for the disposal of surplus pesticides and pesticide containers.

--J. E. D.

AN OVERVIEW OF NON-POINT WATER POLLUTION FROM THE URBAN-SUBURBAN ARENA

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ABSTRACT

Airborne deposition, erosion and corrosion of surfaces, construction areas, septic tanks, urban litter, chemical additives, and animal sources are the fundamental non-point sources of water pollution in urban areas. Information concerning the concentrations, interactions, and movement from these sources into and through the stream system is fragmentary. Observational programs are needed which combine the necessary hydrology with measurements of water quality, stream morphology, and biology so that we can better understand stream quality. Thus, research is indicated which would make it possible to better predict the concentration level of different pollutants that would cause significant impairment of water quality for different water uses.

INTRODUCTION

A search of Water Resources Scientific Information Center abstracts on the subject of non-point sources of water pollution in urban areas indicates that about 500 papers have key words relative to the subject. However, essentially none of these report comprehensive studies in space and time. One reason for the lack of comprehensive investigation may be that the subject has sometimes been referred to as the "secondary" sources of pollution. The secondary connotation may have been a mistake, because the literature reports [Saytor and Boyd, 1972] that storm water pollution loads may be as high as the sanitary sewage from a given urban area. In fact, storm water may contain dangerous levels of toxic materials not ordinarily found in sanitary sewage [Pitt and Amy, 1973]. To be sure, the hydrologic conditions for the two sources are quite different. Sanitary sewage is usually most damaging during dry weather and has considerable persistence, whereas storm water runoff is likely to impact the channel system most intensely during the initial flush of streamflow and lasts only a few hours.

The literature reports on many relatively small, isolated investigations of non-point pollution dealing with special situations. Some important parts of the subject have not been studied because of the difficulty of making meaningful measurements. An example is the movement and alteration of corrosion and weathering products from various structures in urban areas.

As a result of the dearth of comprehensive studies on urban-area non-point sources of water pollution, this paper is mostly a series of introductory and concluding statements in an effort to lend perspective to the subject and form the base for some rather extensive conclusions, which hopefully may incite more comprehensive research and ultimately better control methods.

Understanding the subject

The term pollution seems to trigger considerably different concepts among different people [Haney, 1966]. The pollutor's notion is understandably different from the water user; but within either of these groups there is considerable variation of concept. According to Meier [1971], the conceptual origins of pollution can be traced back to the violation of taboos. Also, the semantic dichotomy of "dirty vs clean" can be considered in the modern society as the basis for the arbitrary taboo relative to pollution. Some environmental belief systems seem to approach religious beliefs, as the natural environment is sanctified and violations of "taboos" become serious crimes. Meier further indicates that this popular worship of nature may damage the rest of society whenever freedom of choice is limited and delays in construction of needed public facilities occur. His conclusions are that a strong commitment to the simple life will only change the location of existing pollution, and that policy makers and planners must seek ways to avoid conflicts and postponements of important work wherever adherence to the modern taboo is strict or excessive.

A balanced perspective on water pollution would seem to require awareness of the intensity-frequency distribution of the movement of the pollutant into and through the stream as well as the water-use requirement in the stream system. That is, the impact of different pollutants can change dramatically with time and with distance along the stream. Perks [1970] suggests that pollution be viewed more as a component of a resource management system than as a control problem. In general, he says that it is essential that the problem of water pollution should not be separated from the other problems of water management--water supply, flood control, recreation, fish and wildlife, and parks and open space.

Information from sound research can be useful to help maintain essential goals and perspective regarding pollution. The benefits of some kinds of pollutants are obvious and yet too often ignored. For example, Boyle [1971] reports that the central part of the Hudson River has excellent fish productivity because the river is nutrientially rich from the populations of Albany and Troy and nearby farmland runoff. In contrast, the upper reaches of the Hudson, which flow through nearby virgin forest, are severely unproductive of fish.

Urban areas and growth

In considering the impact of non-point pollution in urban areas, it is important to note the relative area and complexity of these areas [McGriff, 1972]. Approximately three-fourths (152 million) of the population of the contiguous 48 states is urban [U.S. Bureau of Census, 1973]. Approximately 140 million people live in 264 "Standard Metropolitan Statistical Areas," (SMSAS), each having 50,000 or more persons.

In relation to land use and urban development, it is necessary to point out that the SMSAs include whole county areas in which the population centers are located. Thus, they include a substantial number of rural people and 13 percent of the land of the contiguous 48 states [Council on Environmental Quality, 1973]. Urban uses account for only about 10 percent of the land within the SMSAs; thus, nearly 140 million people occupy only 1.3 percent of the area of these 48 states. Weibel, Anderson, and Woodward [1964] report that 11,400 sewered communities have an area of 43,100 square miles [112,000 sq/km) or 1.2 percent of the area of the country. At that time community population density averaged 2,200 persons per square mile (850 persons per km^2). It is interesting also to note that the largest category of land in the SMSAs in 1970 was forest woodland (30 percent) which reflects to some extent the abandonment of farm land in the SMSAs [Wolman, 1967].

Although the urban areas occupy only a very small part of the United States, they are areas of intense industrial and economic activity and consume vast quantities of energy and resources. In addition, they are undergoing changes within, as well as rapid outward expansion. Expansion results from increases in population and the use of more land per unit of population. The high rate of energy and resource consumption and land-use expansion results in some intense problems with water pollution from non-point sources.

Hydrologic-geologic-biologic interfaces

It has been well demonstrated that urbanization disrupts the hydrologic cycle, thus creating many problems with the quality and distribution of water. Imported water is mostly for transport of waste material and irrigation; but, perhaps the most striking disruption is the increase in rate and amount of overland runoff resulting from a considerable area of impervious surfaces.

Potential movement and alteration of pollutants in a drainage area depends on a host of active and passive agents. The driving forces of movement and alteration, such as solar energy, wind, raindrops, runoff, animals and, by no means least, man, are resisted by the forces of friction and adhesion. Gravity might be considered as both a driving and resisting force working in association with the other forces. Many of the driving forces are lumped under the term, "climate," which on a short-term basis is made up of the strongly unpredictable weather elements. Though urbanization does alter the local weather and climate somewhat, the most dramatic alteration occurs from the morphologic and biologic changes generated by the development of the city complex.

A prevailing concept of a desirable urban area too often requires a vast amount of soil stabilization for transportation and structures, large impervious surfaces with efficient drainage, rapid drainage from structures, some planned vegetation which can be maintained only by application of chemicals, and structures which too often deteriorate to a host of unknown chemicals. Superimposed on the interface of this urban complex with the natural elements is the fast-living pace of a densely-packed population that uses large quantities of materials and energy.

Perhaps the most prevalent pollutant interacting with water is sediment, especially in and downstream from construction areas. Basically, water and sediment interact at the interfaces of the atmosphere, land, fresh water, and saline water subsystems. Of greatest concern are quality changes of the water resulting from the sediment, activated by precipitation, solar energy, evapotranspiration, wind, and water movement [Grissinger and McDowell, 1970; Oschwald, 1972]. Wherever soil is bared, particles at the surface are subject to detachment and movement by raindrop splash. This rapidly reduces infiltration capacity and enhances overland runoff, thus providing rapid downslope transport of the detached particles. The runoff also provides energy for detachment of particles from the channel boundaries. Sediment particles at the interface, or in transport, may transmit sorbed substances from soil to water, or be an active surface for sorption of dissolved substances from the water.

Suspended-sediment pollution is usually thought of as a turbidity problem. Turbidity causes light to be scattered and absorbed rather than transmitted; therefore, it is an optical property of water. The optical property depends on particles to scatter and absorb light, so particle reflectivity and area projected to a light source, rather than mass, are important. Thus turbidity is an extremely poor measure of suspended-sediment content (mass). Nevertheless, it is an important measure of water quality wherever light penetration is necessary for photosynthesis and production of materials in the aquatic food chain, and wherever the water body has aesthetic aspects.

THE SOURCES

Fundamentally, the non-point sources of water pollution from urban-suburban areas can be reduced to seven items each of which may impart varying amounts of particulate and dissolved substances to water. These are:

1. Airborne deposition, including dustfall and material in precipitation. An example, among many thousands, might be asbestos from brake linings.

2. Erosion and corrosion products from surfaces, including wear from tires and streets, deterioration of roofs and buildings, and erosion from soil surfaces.

3. Construction areas, including sediment from exposed soils, chemicals from oil fuel, solvents, stabilizers, paints and litter from packaging and construction discards.

4. Septic tanks can be a problem, especially where permeability is inadequate and surface contamination results, or where permeability is too rapid and ground water contamination results.

5. Urban litter, including miscellaneous trash, lawn, and garden debris and anti-skid materials such as sand or cinders.

6. Chemical additives, including pesticides, herbicides, fertilizers and soil conditioning, ice control and, by no means least, petro-chemicals from transport equipment.

7. Pets, wild animals, and birds.

It is difficult to understand the intensity distribution of pollutants in the waters of the urban area, and downstream therefrom. The difficulty results from a poor knowledge of the intensity distribution of the pollution sources and how they might be altered in time and space within the urban environment. This means that all pollution sources need to be researched as a unit in order to evaluate chemical changes, sorption-desorption characteristics, and mutual reinforcement on the many kinds of surfaces in the basin and in the water environment. An important question is to determine which pollution sources are likely to overshadow others in their impact on waters. Another is whether it is economically, socially, and technically feasible to effect a solution to a given problem. For example, it is usually desirable that some fertilizer continue to be used to maintain an attractive landscape even though low nitrogen is best for water supplies.

Unfortunately, information is fragmentary concerning the concentration and movement characteristics of materials from the various sources. Airborne disposition has been investigated because of interest in the air as a pollution carrier and its impact on man and the biologic complex. Construction-area sediments have been investigated at a number of locations, but the capability to predict what will happen for a given situation is minimal. The design for septic tank systems has been well-established for several decades, yet numerous failures occur. Work recently completed on the release of soluble phosphorus from leaves [Cowen and Lee, 1973] shows (1) that shredded leaves release three times as much phosphorus as unshredded leaves, and (2) that adequate leaf pickup is important. Chlorides applied for ice control have been measured in streams at many locations, but the degree of their impact has not been defined. Programs to define the impact of other landscape chemicals on natural waters seem lacking.

Excretion of pets as a source of pollution in water is an unlikely candidate for study, mainly because pets are a part of the family. On the other hand, studies are likely where bird population is excessive. Loehr [1969] reports average daily manure production, as a percentage of body weight, of 8, 6, 6, and 5 percent for dairy cattle, beef cattle, hogs and laying hens, respectively. If our well-fed pets could be expected to yield 6 percent, and if average pet weight per household were 10 pounds, then the expected daily manure yield from pets might be 0.6 pounds per household. If BOD, total dry solids, and total nitrogen average about 0.002, 0.010, and 0.005 pounds per animal pound, respectively, then average daily pollution potential per household might be 0.02, 0.10, and 0.05 pounds of BOD, total dry solids, and total nitrogen. On an annual basis this is 7, 36, and 18 pounds, respectively--quite enough for considerable concern.

STREET LITTER AND STORM WATER RUNOFF

Though a discussion of the amount of the particulate and dissolved substances from the sources outlined above is not possible, studies of litter, street sweepings, and storm water analyses may be helpful. Table 1 [American Public Works Assoc., 1969] compares the estimated quantity of dustfall, household refuse, street sweepings, and catch-basin material with domestic sanitary wastes on a monthly basis for a typical 10-acre (4-ha) residential area in Chicago. It would be expected that most of the seven tons of dustfall for the

TABLE 1

Monthly Tonnage of Identifiable Solids from a Typical 10-Acre (4 ha) Residential Area, Chicago, Illinois

Month	Air pollution dustfall	Household Refuse		Exterior solids		Total*
		Garbage	Rubbish	Street sweepings	Catch basins	
JAN	0.68	1.8	7.2	0.68	0.43	13.2
FEB	0.78	1.8	6.8	0.68	0.43	12.9
MARCH	0.80	2.3	7.9	0.76	0.43	14.6
APRIL	0.60	2.3	8.8	0.76	0.43	15.3
MAY	0.72	2.3	8.5	0.76	0.43	15.1
JUNE	0.40	2.8	9.2	0.76	0.43	16.0
JULY	0.46	2.8	8.7	0.76	0.43	15.6
AUG	0.30	2.8	9.3	0.76	0.43	16.0
SEPT	0.45	2.3	8.9	0.76	0.43	15.3
OCT	0.56	2.3	8.3	1.56	0.43	15.5
NOV	0.61	2.3	8.4	1.56	0.43	15.6
DEC	0.66	1.8	8.2	0.68	0.43	14.2
Total	7.00	27.6	100.2	10.50	5.20	179.3

* Total includes an average of 2.4 tons/month from sanitary wastes.

year would move off the area with storm runoff, that a very low percentage of the household refuse would reach the storm water and that a substantial amount of the type caught up as street sweepings might be included in storm water because it is not possible to sweep immediately prior to each rain. Further, storm water will contain substantial quantities of dissolved materials not considered part of "identifiable solids."

Street sweepings in Table 1 were 4 percent rags and paper, 64 percent dust and dirt, 22 percent vegetation, and 10 percent inorganic. Three-fourths of the vegetation was picked up in October and November. It was noted that 5.2 tons of material retained by catch basins from the 10 acres (4 ha) consisted of most of the components found in street sweepings of the same area.

Monthly dustfall at 20 stations in Chicago ranged from 21 to 61 tons/mi^2 (8.2 to 23.7 g/m^2) during 1966 with an average monthly distribution about like that found for the 10-acre (4-ha) site listed in Table 1. For comparison, it is of interest to note that the average monthly suspended-sediment yield of the

TABLE 2

Estimated Average Concentration of Dustfall Pollutants in Impervious-Area Runoff

	Dustfall, T/mi^2		Annual precipitation	Impervious area runoff	Runoff-weighted concentration
	(month)	(year)	(inches)	(inches)	(mg/l)
New York	68.3	766	42	34	314
Cincinnati	22.3	268	40	32	115
Pittsburgh	51.1	613	36	29	292
Seattle	25.5	306	39	31	136

TABLE 3

Average Monthly Dustfall Components in mg/m^2/mo. From 46 Sampling Sites for Seattle Vicinity, 1964.

	June	July	August
Total dustfall	8600	6900	7900
Total insoluble	4400	4200	5000
Volatile	1590	1280	1440
Ash	2810	2930	3580
Total soluble	4200	2700	3000
Volatile	-	1280	1150
Ash	-	1390	1690
Calcium	290	323	272
Nitrates	33	72	48
Florides	1.9	2.1	3.2
Sulfate	1460	1280	1180
Phosphate	9.3	8.1	13.4
Lead	13.0	12.0	14.2

Potomac River Basin is about 17 tons, including the effect of the urbanizing areas of metropolitan Washington, D.C. Johnson, Rossano, and Sylvestor [1966] report dustfall data in seven cities, mostly for the 1950's, to range from 15.5 to 63.8 tons/mi^2/mo (6.0 to 24.7 g/m^2/mo). These data for New York, Cincinnati, Pittsburg, and Seattle were used to estimate the average annual concentration of the dustfall in runoff from impervious areas [Table 2]. The assumption was made that the impervious-area runoff was 80 percent of the mean annual precipitation.

Johnson, et al. [1966] states:

> "Dustfall is a measure of the settleable particulate matter (normally in the range 20 to 40 micrometers in diameter) that falls out of the atmosphere because of the influence of gravity. Also included in dustfall would be gaseous, liquid, or solid contaminants soluble in rainwater or snow and washed out of the atmosphere. The quantity, chemical composition, and size of the particulate matter deposited in a given location can vary widely depending on topography, number of local sources, wind patterns and extent of rainfall.... A good part of the dustfall that settles over a city is water soluble and can gain access to any open body of water such as lakes and reservoirs. This water soluble fraction of dustfall can have a significant effect on drinking water quality as well as provide nutrients for micro-organisms in the water."

Johnson, et al., also report extended analysis of dustfall samples from about 40 locations in Seattle during June, July, August, 1964. The average quantities of ash and volatile material in the soluble and unsoluble components are given in Table 3, as well as soluble calcium, nitrates, fluorides, sulfate, phosphate and lead for each month.

Perhaps the critical combined effect of the various non-point sources of pollution can be obtained through the relatively simple investigation of storm water runoff. Pollutants in storm water runoff would result from the concentration and conditioning of materials from dustfall, precipitation, abrasion and corrosion products, sediments, chemical additives, urban litter and other materials in the basin. Some of the results of storm water studies are given in Table 4.

Weibel, Anderson, and Woodward [1964] found in a study of storm runoff from two drainage basins in Cincinnati that the concentration of constituents in runoff changed little for different seasons. The concentration does change dramatically during the course of individual storms, however, as indicated in Table 5. The results in Table 5 include storms of short duration and high intensity as well as those of longer duration and low intensity. The study indicated that storm water from a residential area contained 140 percent more suspended solids than did sanitary sewage from the same area--volatile suspended solids was 44 percent, chemical oxygen demand was 25 percent, biochemical and total nitrogen was 11 percent, respectively. Cincinnati's air pollution sampling indicated that 506 lb/acre/yr (576 kg/ha) of dustfall may account for a substantial part of the 730 lb/acr/yr (820 kg/ha) of suspended solids in storm water runoff.

TABLE 4

Characteristics of Storm Water

City	BOD mg/l	Total Solids mg/l	Suspended Solids mg/l	Coliform /ml	Chlorides mg/l	COD mg/l
1. East Bay Sanitary District						
Minimum	3	726	16	4	300	
Maximum	7,700		4,400	70,000	10,260	
Average	87	1,401	613	11,800	5,100	
2. Cincinnati, Ohio						
Average	17		227			111
3. Los Angeles County						
Average 1962-63	161	2,909			199	
4. Washington, D.C.*						
Minimum	6		26		11	
Maximum	625		36,250		160	
Average	126		2,100		42	
5. Seattle, Washington	10			16,100		
6. Stockholm, Sweden						
Minimum	17	30		40		18
Maximum	30	8,000		200,000		3,100
7. Pretoria, South Africa						
Residential	30			240,000		29
Business	34			230,000		28
8. Detroit, Michigan						
Minimum	96	301	102			
Maximum	234	914	213			
Criteria for potable water**						
To be filtered				5,000	600	10
Not to be filtered				50		10

*Catch-basin samples during storm

**New York State

Source: American Public Works Association, 1969

TABLE 5

Mean Concentrations of Constituents in mg/l in Urbanland Runoff vs Time

	Time after start of runoff				
Parameter	0-15 min	15-30 min	30-60 min	60-120 min	120 min and over
SS	300	280	190	200	160
VSS	98	69	47	58	38
COD	170	130	110	97	72
BOD	28	26	23	20	12
Total N	3.6	3.4	3.1	2.7	2.3
Soluble PO_4	0.99	0.86	0.92	0.83	0.63

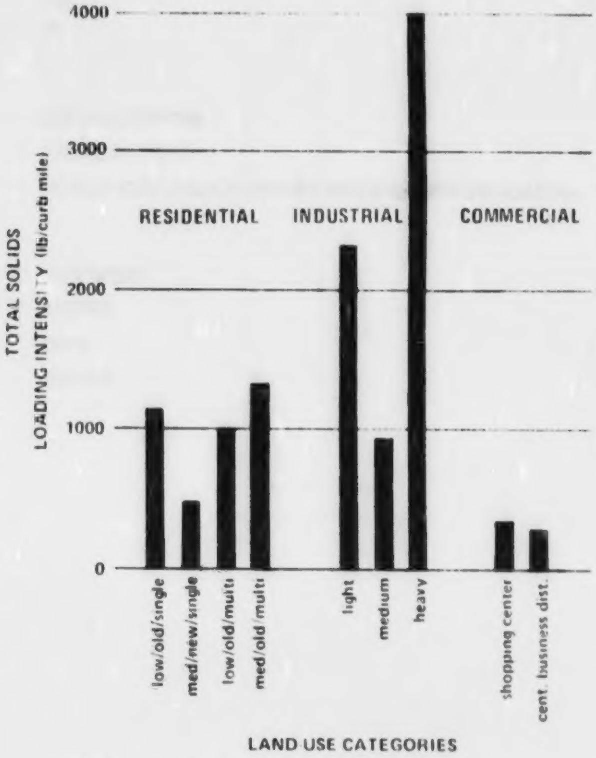
Source: Weibel, Anderson, and Woodward, 1964

A study of pollutional loads of storm water runoff for 15 test areas in Tulsa [Avco Economic Systems Corp., 1970] indicates the difficulty of finding variables suitable for measurement that can be used to predict the impact of non-point pollution. In this study, input variables such as precipitation (current and antecedent), environment (general sanitary conditions), and the geomorphological characteristics of the basins (area, length of channel, relief, and land slope) were correlated with output variables such as bacterial, organic, nutrient, and solids concentrations. Analysis by multiple regression indicated that significant correlation coefficients between input and output variables were few. However, total coliform increased significantly with both relative street area and residential density, chemical oxygen demand increased with residential density, and nitrogen and suspended solids increased with relative street area. In other words, efforts to reduce residential density and street area should help to reduce stream pollution.

A mobile rain simulator capable of covering a 6 by 9 meter area was used to produce street runoff which was analyzed for pollutants [Saytor and Boyd, 1972], at San Jose, Phoenix, Milwaukee, Bucyrus, Baltimore, Atlanta, Tulsa and Seattle at representative residential, industrial, and commercial locations. For all

FIGURE 1

Total Solids Loading on Street Surfaces: Variation with Land Use [from Saytor and Boyd, 1972]



land uses, total solids loading ranged from about 400 lb/curb mi (292 kg/curb km) at Tulsa to about 3,500 lb/curb mi (2,560 kg/curb km) at San Jose. The average total solids loading for all cities is graphed by land-use categories in Figure 1. Average BOD and COD loading for all cities is graphed by land-use categories in Figures 2 and 3. The effect of land use on nutrient (phosphates, kjeldahl nitrogen, nitrates) concentration and loading intensity is shown in Figure 4. The effect of land use on heavy metal loading (chromium, nickel, lead, copper, zinc and mercury) is shown in Figure 5. The results were translated into pounds of contaminant per curb mile per day by land-use category for all cities (Table 6). Table 7 gives the fraction of these pollutants associated with each particle-size range and shows that the phosphates, lead, dieldrin, and DDT are mostly associated with small sizes. The efficiency of removing these small sizes by street sweeping is low. In fact, the table shows that efforts to remove particles larger than 2 mm from streets by sweeping will generally remove no more than 10 percent of many types of pollutants, even if the removal of the part more than 2 mm were 100 percent effective. Between 40 and 90 percent of the pollutants are associated with particles less than 1/4 mm in diameter.

FIGURE 2

BOD Loading Intensities on Streets:
Variation with Land Use
[from Saytor and Boyd, 1972]

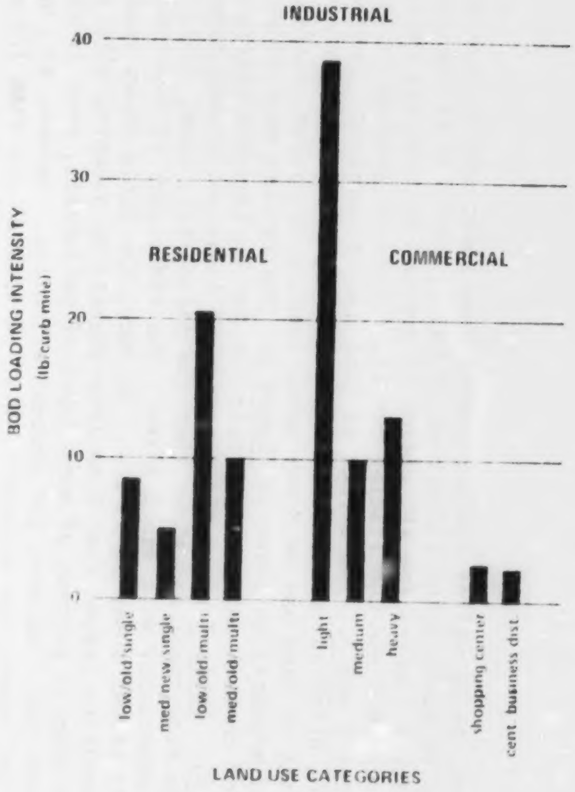


FIGURE 3

COD Loading Intensities on Streets:
Variation with Land Use [from Saytor and Boyd, 1972]

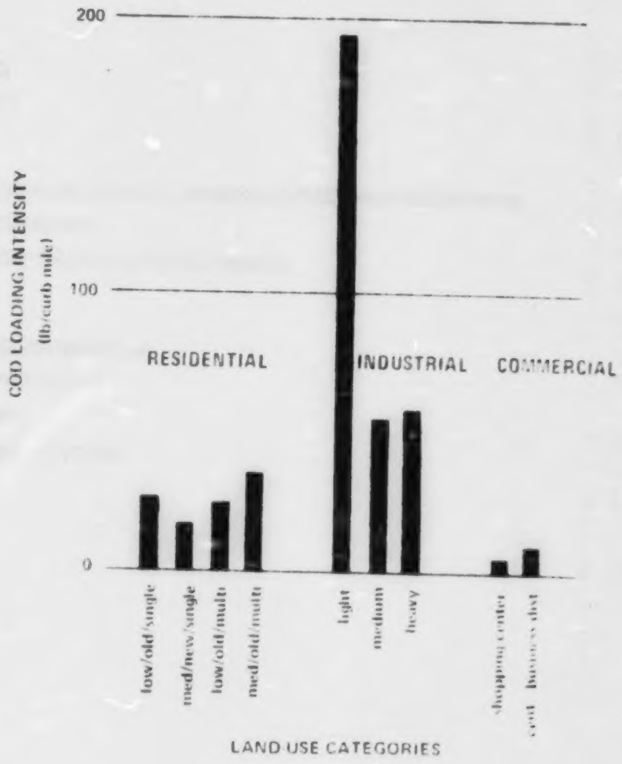


FIGURE 4

Nutrient Loading Intensities and Waste "Strengths": Variation with Land Use [from Saytor and Boyd, 1972]

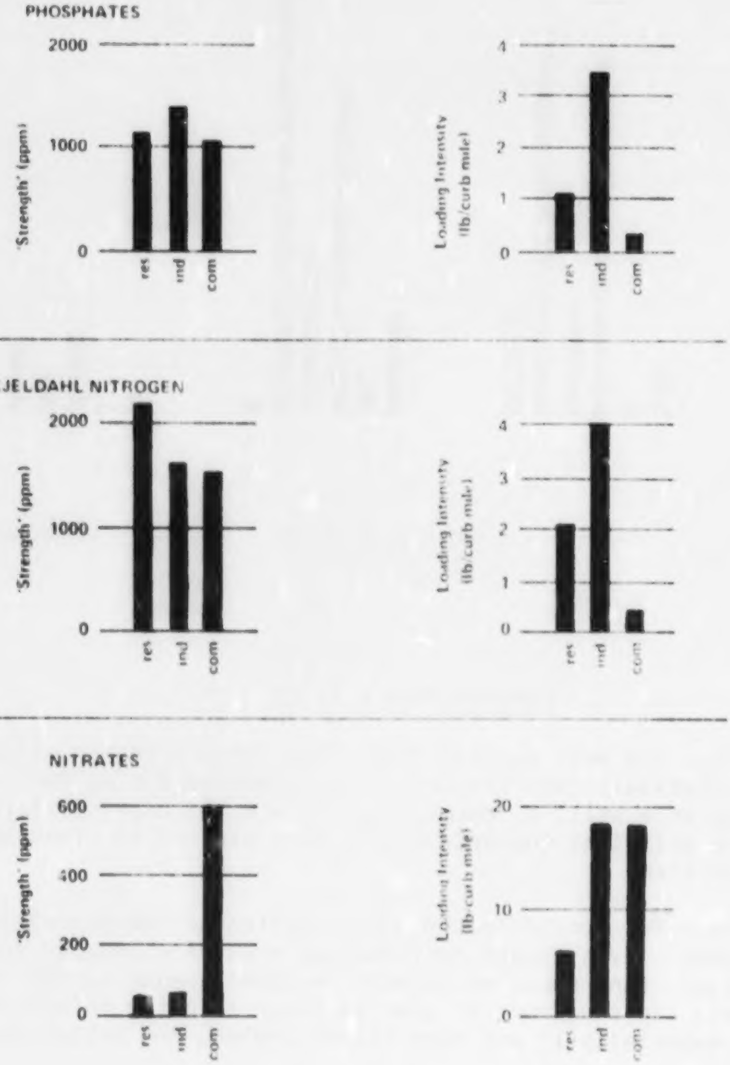
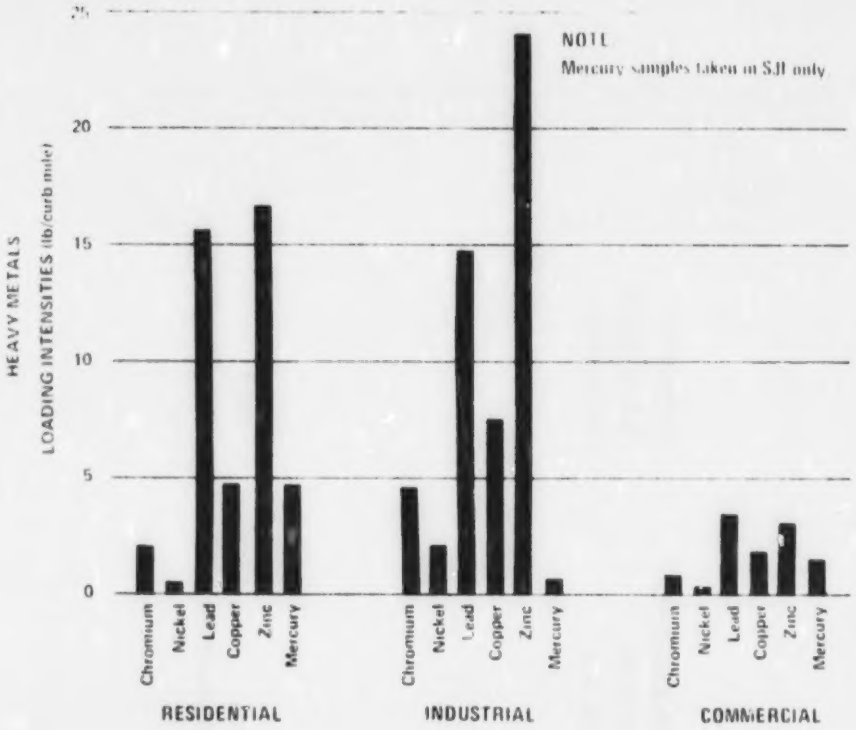


FIGURE 5

Heavy Metals Loading Intensities on Street Surfaces: Variation with Land Use [from Saytor and Boyd, 1972]



CONSTRUCTION AREA SEDIMENT

A few decades ago the main purpose of erosion control was to maintain an organically rich, relatively fine-textured rooting medium for productive plant growth. At that time we were still a "rural nation," and the cause of soil erosion control had strong political support because each user of the land had an important stake in the problem.

Today, the reason for erosion control has shifted to the prevention of sediment impact on streams. Even though construction areas are usually very small relative to crop land, they cause an intense sediment impact on the stream system, especially where the construction area is large relative to the stream's drainage area, and especially in and near cities [Wolman and Schick, 1967].

Larger drainage areas have a wide variety of uncoordinated construction activities by a large number of contractors, and the stream may be impacted for several years on a relatively low intensity of construction. Very small drainage

TABLE 6

Distribution of Contaminant Load by Land-Use Category[1]
(lb/curb mile/day) (From Seytor and Boyd, 1972)

	Residential[2]	Industrial	Commercial
Total Solids	590	1,400	180
Volatile Solids	44	77	14
BOD	3.6	7.2	.99
COD	20	81	5.7
Kjeldahl Nitrogen	0.60	1.2	.12
Nitrates	0.019	.055	.055
Phosphates	0.37	1.1	.10
Total Heavy Metals	1.2	1.6	.34

1. Based on number of days since cleaned, either by rain or by sweeping, whichever occurred the closest to the test date.

2. The figure reported here as "residential" was computed by combining all of the observed data for the four residential land-use categories sampled in each city. "Industrial" and "Commercial" figures were computed similarly.

areas, on the other hand, may be severely impacted for only a few weeks when a single construction site covers nearly all the drainage area. The diversity of responsibility for these construction-area impacts on streams is apparently the primary motivation for some states and counties to require erosion and sediment movement controls in construction areas.

Because of experience in controlling erosion in rural areas, the Soil Conservation Service of the U.S. Department of Agriculture took an active role in the preparation of construction-area erosion-control guidelines [USDA Soil Conservation Service, 1969, 1970]. These made it relatively easy for state and local governments to adopt "erosion and sediment control ordinances" which include rather specific instructions for development of erosion and sediment control guidelines [Virginia Soil and Water Construction Commission, 1974; Pennsylvania Department of Environmental Resources, 1973]. Efforts to reduce erosion during

TABLE 7

Fraction of Pollutant Associated with Each Particle Size Range (% by Weight) [from Saytor and Boyd, 1972]

	PARTICLE SIZE (micrometers)					
	> 2,000	840 - 2,000	246 - 840	104 - 246	43 - 104	< 43
Total Solids	24.4	7.6	24.6	27.8	9.7	5.9
Volatile Solids	11.0	17.4	12.0	16.1	17.9	25.6
BOD_5	7.4	20.1	15.7	15.2	17.3	24.3
COD	2.4	4.5	13.0	12.4	45.0	22.7
Kjeldahl Nitrogen	9.9	11.6	20.0	20.2	19.6	18.7
Nitrates	8.6	6.5	7.9	16.7	28.4	31.9
Phosphates	0	0.9	6.9	6.4	29.6	56.2
Total Heavy Metals	16.3	17.5	14.9	23.5	27.8	
Chromium	26.1	13.6	16.3	16.3	27.7	
Copper	22.5	20.0	16.5	19.0	22.0	
Zinc	4.9	25.9	16.0	26.6	26.6	
Nickel	26.2	14.2	15.3	17.2	27.1	
Mercury	16.4	28.8	16.4	19.2	19.2	
Lead	1.7	2.6	8.7	42.5	44.5	
Average	16.3	17.5	14.9	23.5	27.8	
Total Pesticides	0	16.0	26.5	25.8	31.7	
Dieldrin	0	.9	21.3	36.0	41.8	
PCB	0	33.5	32.4	15.7	18.4	
DDD	0	10.7	29.3	30.3	29.7	
Methoxychlor	0	35.0	27.0	11.0	27.0	
DDT	0	0	22.3	36.1	42.1	

construction of a large office complex at Reston, Virginia, did reduce sediment movement in a small stream draining the area [Guy, 1974]. Whether or not the value of the cleaner stream was worth the cost of preventing the sediment movement is quite an open question. A report by Yorke and Herb [in preparation] evaluates sediment-control methods used in association with construction in the Rock Creek and Anacostia Basins, Maryland, from 1966 to 1974.

A substantial part of the difficulty in maintaining effective and worthwhile sediment-control programs in construction areas involves the rapidly changing environmental conditions, which cause the potential for sediment movement to and in streams to change rapidly. These man-made changes are superimposed on natural during-storm and seasonal changes [Mansue and Anderson, 1974; Vice, Guy and Ferguson, 1969]. Therefore, analysis of potential sediment movement in a stream is usually generalized on an annual basis through a "universal soil loss equation." This indicates gross erosion which is adjusted for the effects of sediment deposition between the area of erosion and the stream location of interest as shown by the following equation [Wischmeier and Smith, 1965; Guy and Jones, 1972]:

$$S_y = S_d \; R \; K \; LS \; C \; P$$

In this equation, S_d is the ratio of the total expected erosion in the basin to sediment transported from the basin [Roehl, 1962]; R is a rainfall factor expressing the erosion potential of average rainfall in the locality [Wischmeier and Smith, 1958]; K is the erosivity or average soil loss in tons per acre per unit of R from a completely exposed soil on a 9 percent slope 73 feet long [Wischmeier, Johnson and Cross, 1971]; LS is the topographic factor for slope gradient and slope length; C is a cropping management (vegetative cover) factor; and P is an adjustment for the effects of applied erosion-control practices.

Though the above equation gives expected sediment yield on an annual basis, techniques are available [Wischmeier and Smith, 1965; Chen, 1974] to make estimates on either a monthly or storm-event basis. Chen provides a procedure, with the aid of graphs, by which the planner can quickly choose needed alternative methods for erosion control to prevent sediment movement in the stream from exceeding some maximum limit.

Basically, a sound sediment-control system should begin with land use and structure design that minimizes sediment problems during construction, then makes use of practical biologic controls that prevent erosion on as much of the site as possible, and then makes use of physical controls to slow runoff and trap sediment before it reaches important waterways. The physical controls do not effectively control the movement of fine sediment; therefore, it is important that as many biological controls as possible be used.

The most lasting impact of many construction projects on storm runoff results from paving and roofing over a part of the drainage basin. This, in turn, increases the amount and peak rate of flow in the stream which, again in turn, causes the stream to enlarge the channel by erosion to accommodate these increased flows. The enlargement may take several years; thus, the process causes a long-term sediment impact on the stream.

DISCUSSION

It has been said in many different ways that pollution is the impairment of water quality with respect to some beneficial use. This suggests a need to determine the level of pollution at which water becomes impaired for various uses. Obviously, the matrix necessary to answer this question is extremely complex when potential pollutants and beneficial uses each number in the thousands. When the solution to the matrix is largely incomplete, as it is today, we can use many simplifying assumptions and live with approximate answers; or we can adopt the "big gun approach" and spend a vast amount of effort on "stopping" pollution, with the result that we may kill business, or with the result that the effort is largely ineffective.

This paper outlines some aspects of seven fundamental non-point sources of water pollution in urban areas--airborne deposition, erosion and corrosion from surfaces, construction areas, septic tanks, urban litter, chemical additives, and animals. Information concerning the concentration, interactions, and movement from these sources into and through the stream system is fragmentary. Especially lacking is information on the impact from erosion and corrosion from surfaces, construction areas, and animals.

Even where some information has been obtained, the impact of non-point pollution on water resources reported in the literature seems to be based on sketchy data. Laboratories have a vast array of relatively expensive equipment for making sophisticated analysis of samples for whatever we are looking for. The shortcoming seems to involve the sampling scheme--in other words, the old problem of obtaining a representative set of samples (1) in space and time, or (2) of the flow of the stream. In many situations, especially in dealing with these non-point sources, it is necessary to collect a large percentage of the samples during storm runoff. This usually involves working in the rain, which is quite enough to dampen the spirit. Consequently, it is easier to obtain samples that are biased to dry weather or recession-flow conditions. It also is evident that systems analysis and development of models have advanced well beyond needed data on which to make these models really useful.

Before we go all-out to remove all "pollutants," we need to keep in mind that nature always has been, is now, and always will be a very heavy "polluter." The late Dr. W. T. Pecora, former director of the U.S. Geological Survey, reminds us that rainfall pours more than 4 million tons of sodium chloride, 2 1/2 million tons of sodium sulfate, and 26 million tons of calcium compounds on the United States each year. Just three volcanic eruptions in the past 90 years have injected more dust, ashes, and gasses into the atmosphere than has all of mankind since the beginning of time [Gould, 1973].

Wolman [1971] has suggested that "few observational programs combine the necessary hydrology with measurement of water quality, river characteristics, and biology." In other words, hydrology rather than water quality is needed to evaluate river quality. He states further that interpretation of present data requires that the analyst be familiar with both the data and with the changing characteristics of land use and economy of the basin. Our measurement of a multitude of parameters at selected problem areas may shed little light on the dynamics of the processes active in river systems. As indicated, it seems desirable that additional attention be given to the measurement of parameters related to river behavior and to input-output analysis of environmental material, man-made and natural. Lastly, it seems evident that the goal of better data systems is to make it possible to better judge the effectiveness of committing large sums of money; which in turn, requires that the observational tools be designed to evaluate the response of the rivers to expenditures that have already been made for this purpose.

Available data on non-point water pollution indicates that the impact from runoff can be reduced considerably if runoff were better managed. To manage runoff better would require that some administrative codes and design criteria be changed in urban development. For instance, some methods of storm-runoff management to reduce pollution include:

1. The use of temporary storage of precipitation on roofs. This will reduce contamination not only from dustfall but also from the weathered products of the roof.

2. Where roof storage is not practical, letting the drainage flow over the lawn, through a subsurface area of a garden, or at least into a dry well.

3. Eliminating curb loading and "high runoff efficiency" by substituting vegetated drains for the curb-drainage system. This can easily be accomplished in most single-family residential areas.

4. Providing temporary storage on parking lots and lawn or other open-space areas.

It is self-evident that these methods would generally increase the cost and amount of attention required for maintenance. However, this increase in maintenance cost is sometimes offset by greatly decreased capital costs--for example, the elimination of curbs and underground storm drains. It is also evident that drainage areas could be retained as open spaces, indicating that better land use planning would be necessary.

The review of pertinent literature indicates that storm water runoff is sometimes the most feculent water to reach our streams, lakes and reservoirs, many of which are in or near urban areas. The above examples of methods for storm runoff management emphasize the thesis that water pollution needs to be viewed as an important component of water resource management, rather than as a control problem.

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NON-POINT POLLUTION

FROM MINING AND MINERAL EXTRACTION

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INTRODUCTION

Mining can be divided into two categories: surface and underground. Surface mines can further be divided into three basic types: (1) area mines, located on relatively flat land, usually less than 200 feet deep and covering large areas, (2) contour mines, found in mountain areas, usually less than 150 feet deep, narrow and long, and (3) pit or quarry mines, usually deep, often having a high mineral-to-overburden ratio and mined on benches. Non-point pollution from surface mining includes sediment, acid mine drainage, and highly mineralized water.

Underground mines often encounter water which must be removed by either pumpage or gravity drainage. Since this water must exit the mine either through a pipe or channel, it would generally be considered a point source and thus must be treated as laws and regulations require. Most of the discharges from surface mines, since they, too, are pumped or flow through control structures such as sediment ponds, can be considered point sources.

Both surface and underground mines that discharge upon abandonment may be considered non-point sources, primarily from the standpoint that no one is held responsible. Since the 1930's, it has been recognized that abandoned underground mines were the major source of acid mine drainage. The magazine Coal Mining [Anon., 1938] reported that 18.7 million pounds of acid were being discharged from coal mines. Of this figure, almost 55 percent,or 10.3 million pounds, were being discharged from abandoned mines.

The Appalachian Regional Commission Report for 1969 presented figures for the coal mines within the Appalachian area. As these figures show [see Table 1, next page], over half of the acid pollution was from inactive underground mines. In addition, most of the acid being discharged from inactive combined sources was originating from underground mines. Thus approximately 60 percent of the acid problem was directly or indirectly related to abandoned underground mines.

TABLE 1
Acidity Figures for Appalachian Area Coal Mines
(Source: Appalachian Regional Commission Report, 1969)

Source	Acidity 1,000 lb/day	Percent
Underground, Active	614	19
Underground, Inactive	1,712	53
Surface, Active	28	01
Surface, Inactive	361	11
Combined, Active*	60	02
Combined, Inactive*	238	07
Other	245	07
	3,258	100

*Includes sources where underground could not be separated from surface.

UNDERGROUND MINES

Since acid formation has been found to be a result of the oxidation of pyrite, all preventive technology is based on the segregation of pyrite from air. Water serves as a transport media and a reactant in the oxidation process. Water-control methods attack the transport phases and not the reactant phase, since sufficient water is available in the humid atmosphere of an underground mine to satisfy the oxidation process. Bacteria have been implicated as a catalyst in the oxidation of pyrite. Although the importance of bacteria is still under debate, some efforts have been made to control bacteria. A study was supported by EPA [Shearer and Everson, 1969] to evaluate the use of bacteriophage. The results of this study produced no evidence that bacteriophage were either available or able to control the formation of acid mine drainage. Generally, antibacterial agents, even if found effective, are not considered a viable answer to the underground acid mine drainage because of the difficulty in applying the material throughout an underground mine and the probability of having to apply the material on a periodic basis. The bacteria associated with mine drainage formation are abundant in soil and water, and reinoculation of the mine area would occur rapidly, necessitating retreatment of the area.

Air Control

Ever since the 1920's, when it was documented that pyrite oxidation was the cause of acid mine drainage, attempts have been made to prevent air from entering underground mines. The massive mine-sealing program of the 1930's is but one example. Air control has been accomplished basically by one of three methods: (1) sealing, plugging, filling, or closing off all portals, boreholes, openings, cracks, fissures, etc., to prevent air from entering the mine working; (2) filling the mine working with water as an oxygen barrier, and (3) filling the mine working with an oxygen-free atmosphere.

The debate over the effectiveness of placing an air seal in mine portals and sealing known openings into an underground mine has been waged for years. It has been shown many times in laboratory studies that if oxygen is excluded from the mine, acid mine drainage formation will cease. The major problem lies in actually sealing an underground mine so that the oxygen level is reduced sufficiently to cause a significant decrease in acid formation. Presumably the oxygen concentration must approach 1 percent to obtain a better than 90 percent reduction in acid formation. Two of the best documented studies, one by the U.S. Bureau of Mines [Moebs and Kieckovic, 1970] and one by EPA [Hill and Martin, 1972], showed that even where extraordinary efforts were taken to seal the mine, the reduction of acidity was only 50 to 60 percent. Part of the reduction was accounted to the changing of the hydrology of the mine and preventing water from entering the working, and the remainder to reducing the oxygen content within the mine. The oxygen content within the Bureau of Mine's mine was never reduced below 14 and averaged 17 percent, while the EPA study produced a low of and average of about 15 percent. It is obvious from these figures that air still had access to the mine. The source was probably the cracks, fissures, and fractures in the overburden material. Based upon the 15 to 17 percent oxygen in the mine, it is difficult to explain a 50 to 60 percent decrease in acid production.

In summary, it is our feeling that the effectiveness of a first-class, air-sealed mine--with all known openings, subsidences, and the like sealed--is about 50 percent. Mines with shallow cover, outcrops surface mined, and mines highly subsided would be less conducive to air sealing. An air-sealed mine requires maintenance to assure the integrity of the seal.

Several investigators have noted that when pyrite is submerged under water, pyrite oxidation essentially ceases. Mines below drainage that are permanently flooded do not normally have an acid mine drainage problem. In recent years, efforts have been made to flood mines above drainage by utilizing bulkhead seals. EPA has sponsored several studies in the development of these seals and in determining their effectiveness. A recent EPA [1973] publication describes the various bulkhead seals that have been tested. One of these has been called the permeable limestone seal. Limestone is placed in the mine portal and sealed to the walls and roof with grout. As the acid mine drainage passes through limestone, an iron hydroxide and calcium sulfate precipitate is formed which fills the pores in the limestone, producing a solid bulkhead-type seal. In addition, water that passes through the limestone before it is sealed is neutralized. Although this seal was developed in 1968, EPA has recently concluded a laboratory study [Penrose and Holubec, 1973] on optimizing the design of the seal, and currently is utilizing this design to seal a mine near Morgantown, West Virginia.

An integral part of all bulkhead seals is a grout curtain adjacent to the seal. The driving of a portal results in the cracking of the material adjacent to the portal, and these cracks become a weak point in the system. It is often in the outcrop adjacent to the seal, and not the seal itself, where leaks occur. Grout curtains are installed to seal this area.

Bulkhead seals have been used with heads up to 35 feet. The effectiveness of this seal depends on the integrity of the outcrop, the amount of the mine working that is permanently flooded, and the soundness of the seal. Effectiveness has ranged from 100 percent, where there is no longer a discharge, to as low as zero--the latter when a working extended so close to the outcrop that a barrier could not be established.

In summary, we feel that bulkheading is one of the most effective techniques available for controlling pollution from abandoned underground mines. It has been proven for heads up to 35 feet and may be applicable to higher heads. Where a sound outcrop is available and all of the working can be permanently flooded, the expected effectiveness is greater than 95 percent. Bulkhead-sealed mines should have regular, preferably annual, inspection and maintenance because of the water impounded.

EPA [1971] has evaluated the concept of filling a mine with inert gas. Laboratory studies have shown the potential effectiveness of this method. A field study [Robbins, 1973] demonstrated the difficulty of sealing a mine to the extent that excessive gas loss would not occur. Gas is lost through cracks, fissures, fractures, and the like, and oxygen enters by the same path with each change in barometric pressure. Filling a mine with inert gas is similar to treatment of the discharge in that it would be a continuing process requiring almost daily operation and maintenance. For this reason, inert gas is not considered a solution to the abandoned-mine problem.

Water Control

The basic approach to water-control methods is to prevent water from entering the mine working, where it could flush and transport the products of pyrite oxidation from the mine. These methods do not have a major effect on the pyrite oxidation process itself. Since it would be impractical and prohibitively expensive to prevent all water from entering the mine, only major water sources usually are controlled. Thus the method cannot be 100 percent effective.

Major sources of water directed to underground mines that can be controlled are: (1) streams that have been diverted to underground working or lost by way of subsidence holes, fractures, etc.; (2) surface mines that trap and direct water to underground mines, and (3) fractures, fissures, and cracks extending into the mine. EPA has been active in demonstrating control methods for streams and surface mines, but has found the control of fractures and other flaws difficult because it is hard to locate and seal these openings. Measuring the effectiveness of water-control techniques has been difficult. Generally the quantity of water diverted from underground mines can be measured. However, measurement of decreases in mine-discharges flow has been only marginal, because of the long resident time water has in the mine complex and the hydrology of groundwater flow that clouds the picture.

Common techniques used for water control are: (1) grading to facilitate rapid runoff away from the mine; (2) rechanneling of streams; (3) lining of streams; (4) filling and compaction of subsidence holes; (5) diversion ditches, and (6) sealing of boreholes, mine openings, fracture zones, and auger holes that allow water to enter the mine. One attempt was made [Uniroyal, 1972] to develop a latex blanket over an entire mine, but it was unsuccessful.

Another technique discussed in the past few years is aquifer control. Aquifers above and adjacent to the mine are either drained by gravity or pumped through walls to dewater the mine. These techniques hold promise, but have yet to be demonstrated.

In summary, water-control methods appear intuitively to be good ways to reduce acid mine drainage, but their effectiveness has not been well documented. A unit decrease in flow does not necessarily mean a unit decrease in acid load, because acid formation may not decrease. The only decrease may be the amount of fluid flushed from the mine.

Filling and Removal

Other approaches to the underground mine problem have included "fill-it-up," "knock-it-down," or "remove it."

"Fill-it-up" entails filling the voids within the mine. Materials that have been suggested as fillers are sand, coal refuse, fly ash, municipal waste, and waste residues such as sludges from acid mine drainage neutralization plants and SO_2 scrubbers. The major difficulty with filling any mine is moving material to the void. Most parts of an abandoned mine are inaccessable by means of the passages cut during mining. Reopening these passages typically would be either impossible or prohibitively expensive. Thus, the only practical entry is through holes drilled from the surface. These holes are expensive, limit the size of the material that can be injected, and limit the distribution of the material as it enters the void.

In those situations where materials have been placed back into a mine, control of acid mine drainage has not been the purpose. The major effort has been to prevent subsidence or control mine fires. A combination of sand and coal refuse has been used in these situations, generally at high cost. The effect of mine filling on acid mine drainage control has never been determined.

Acid neutralization sludges and fly ash have been placed in underground mines as a means of disposing of waste residues, not for the purpose of controlling acid mine drainage. The materials, where they are alkaline, may neutralize acid water in the mine. They also may coat pyrite surfaces, thus preventing acid formation. On the other hand, there is a danger that the material will flow out a mine opening and, if the mine is very acid, that soluble salts will be leached from the residues. An additional problem is the large volume of residues required to fill a mine.

The "knock-it-down" concept has been proposed from two standpoints. The first is to blast the entire mine, causing it to collapse and thus filling all the voids. This method would have high cost, and probably would result in surface damages. The blasting design and implementation to achieve complete collapse would be difficult because of the pillars, subsidence, and access to the mine voids.

The second method is to blast alkaline overburden down into the mine voids. The premise is that as the acid water flows through the alkaline material, it would be neutralized. This method is being demonstrated in West Virginia by EPA and that state.

The "remove it" concept also is referred to as "daylighting." Where the overburden depth is not too great, the underground mine is stripped out, using standard surface-mining methods. Thus the remaining coal (from 25 to 60 percent) is recovered, and the underground mine is removed. The area then is reclaimed as a surface mine. This method is being demonstrated in Maryland by EPA and that state.

Treatment

Technology is available to neutralize the acid mine drainage discharged from underground mines. Except for a few situations, treatment is not considered a viable solution for abandoned mines because of the long treatment period required and because of high costs. A responsible party is not usually available, and the cost must be born by the government. In a few special cases, it may be desirable to treat the water by reverse osmosis or ion exchange for a potable water supply.

SURFACE MINES

Construction of a road to facilitate mining and prospecting is the beginning of the damage to the ecosystem due to surface mining. Vegetation and topsoil rich in organisms are pushed aside or covered with subsoil and rock. All too often in hilly terrain, the "cut and fills" are not balanced--as "cut" material is pushed over the hillside, more "fill" material must be borrowed, thus expanding the amount of disturbance. Runoff from adjacent land, road surfaces, and berms, if not properly controlled, causes accelerated erosion which produces the first off-site damage. Prospecting often is performed with dozers by ripping the soil surface to reveal outcrops of minerals or fuels. These scars are additional sources of sediment. In some states, control of sediment from the mining operation itself is required, but haulroads and prospectings are not controlled--even though they may contribute as much sediment as the mine itself.

Next in the mining sequence is the clearing and grubbing of trees, brush, and other surface materials from the mine site. In some situations the trees are harvested. At other times the material is windrowed below the mine disturbance to serve as a filter for sediment. Such activities expose areas of bare soil and accelerate erosion.

A massive earth-moving operation then begins. Some states require that the topsoil and/or subsoil be stockpiled for later spreading during final grading. The overburden rock usually is fractured by blasting before being ripped apart and moved by shovels, draglines, dozers, front-end loaders, or scrapers. The mining process loosens the overburden material, making it more susceptible to erosion. In addition, breaking up the consolidated material exposes new surfaces, which subject ions on the overburden surfaces to weathering and leaching. The ionic strength of the water discharging from the area increases. Typical ions to increase in concentration are sulfate, calcium, magnesium, manganese, aluminum, and iron. The fracturing process also may expose pyrite to oxygen, and the resulting oxidation process releases sulfuric acid and iron. The acid accelerates leaching of heavy metals such as copper, zinc, aluminum, and manganese from the overburden rock. These metallic ions and the acid place an additional strain on the water ecosystem.

After exposing the ore body or fuel, appropriate equipment is used to remove it for market. Low-grade mineralized waste material often is associated with the seam being mined. This material can be a major source of chemical pollution and should be specially handled. Following removal of the coal, ore, or other deposit, backfilling of the excavation begins. Here, too, environmental degradation may result from removal. The final process, termed reclamation, involves grading, replacing the topsoil where practical, and revegetation. The practices used during reclamation have a major bearing on the recovery rate of the surface mine/site.

EROSION AND SEDIMENT

Sediment in water reduces light penetration and alters the temperature, which directly affects aquatic flora and fauna. Fish production is hindered because food organisms are smothered, spawning grounds destroyed, and pools filled. Sediment places an additional burden on treatment plants and cost as well as on aesthetic and economic values [McKee and Wolf, 1963]. Sediment deposit in navigable streams must be removed, but costs are high. Streams choked with sediment have reduced carrying capacity and subject to flooding.

The mechanisms of soil erosion by water consist of soil detachment by raindrop and water scoring, and transportation by surface flow. Sediment yield can best be described by the Universal Soil Loss Equation (USLE) which combines the principal factors that influence surface soil erosion by water [Roth, Nelson & Romkens, 1974]. The equation takes the form:

$$A = RKLSCP$$

where: A is the soil loss in tons/acre
R is the rainfall factor
K is the soil erodibility factor
L is the length of slope factor
S is the steepness of slope factor
C is the cropping and management factor
P is the erosion control practice factor

The U.S. Department of Agriculture has prepared a handbook for utilizing this equation [Wischmeier and Smith, 1965].

Rainfall factor R reflects the potential of raindrop impact and runoff turbulence to dislodge and transport soil particles. This factor is dependent upon the rainfall pattern and intensity of a specific locality. The mine operator has no control over this factor.

Soil erodibility factor K reflects the most significant soil characteristics affecting soil erodibility. For surface soils, the soil texture, organic-matter content, soil structure, and permeability are the key parameters. Recently Roth et al. [1974] have shown that, for subsoils low in organic matter and high in clay, the chemical properties of the material--such as its iron and aluminum content--have a major bearing on its erodibility. The K factors for the bedrock material fractured during mining and later weathered remains to be defined. A mining company has limited control of this material during the mining operation, but can take advantage of the soil erodibility during reclamation by making every effort to place the material with the lowest erodibility factor on the surface. This material should have a high organic-matter content, high permeability, and be granular. In many situations, the practice of top soiling is a step toward obtaining a lower K factor, provided the soil is properly handled and placed.

Length of slope factor L reflects increased sediment detachment and transport as runoff velocities and volume increase with increased slope length. Spoil should be stored and final-graded to prevent long slopes. Where long slopes must be left, control structures should be installed to break up the slope (see P factor).

Degree of slope factor S also recognizes that runoff velocity increases as the slope of the land increases. For example, increasing the steepness of a slope from 10 to 40 percent doubles the flow velocity. Steep slopes should be discouraged in mining operations. Slopes of less than 33 percent are recommended, but in mountain-type surface mining the original slope may equal or exceed 33 percent.

Cover management factor C reflects the influence of the type of vegetative cover, seeding method, soil tillage, disposition of residues, and general management level. The C value ranges from near zero for excellent sod to 1.0 for bare soil. Mining firms can take advantage of this factor by establishing a quick cover of grasses and legumes and using proper tillage techniques. The C values decrease as the percent of ground cover and the percent of canopy cover increase, and the height of the canopy decreases. The application of mulches such as straw, woodchips, and chemicals also decreases the C factor.

Practice factor P accounts for erosion-control structures such as terraces and diversions. These practices are used to influence the drainage patterns, runoff concentration, and runoff velocity. Numerous such practices are available, and the mining company should take advantage of those applicable to their situation.

A study by Collier et al. [1964] showed that the average annual sediment production from a surface-mined watershed was 42 tons/acre, more than 1,000 times higher than an unmined watershed. Curtis [1974] studied three watersheds in Kentucky and found the sediment yield to range between 0.84 and 1.27 area-inches of area disturbed. He could find little correlation between sediment yield and the amount of land disturbed and concluded that methods of mining and handling the overburden are major factors controlling sediment yield. He also noted that the highest sediment yields were measured during the first six-month period after mining. This indicates the need for more attention to activities during and immediately following mining, and the importance of establishing an adequate cover of vegetation and control structures as quickly as possible after mining ceases.

ACID MINE DRAINAGE (AMD)

The removal of overburden often exposes rock materials containing pyrite (iron disulfide). As shown in Equations 1 and 2 [see Equations following], the oxidation of pyrite results in the production of ferrous iron and sulfuric acid. The reaction then proceeds to form ferric hydroxide and more acid, as shown in Equations 2 and 4.

$$2\,FeS_2 + 2H_2O + 7O_2 \longrightarrow 2FeSO_4 + 2H_2SO_4 \qquad (1)$$

$$\text{(Pyrite)} \longrightarrow \text{(Ferrous Iron)} + \text{(Sulfuric Acid)}$$

$$FeS_2 + 14Fe^{+3} + 8H_2O \longrightarrow 15Fe^{+2} + 2SO_4^{-2} + 16H^+ \qquad (2)$$

(Pyrite) + (Ferric Iron) (Ferrous Iron) + (Sulfate) (Acid)

$$4FeSO_1 + 2O_2 + 2H_2SO_4 \longrightarrow 2Fe_2(SO_4)_3 + 2H_2O \qquad (3)$$

$$Fe_2(SO_4)_3 + 6H_2O \longrightarrow 2Fe(OH)_3 + 3H_2SO_4 \qquad (4)$$

The amount and rate of acid formation, and the quality of water discharged are a function of the amount and type of pyrite in the overburden rock, ore, and coal, time of exposure, characteristics of the overburden, and amount of available water [Moth et al., 1972]. Crystalline forms of pyritic material are less subject to weathering and oxidation than amorphic forms. Since oxidation by oxygen is the primary reaction during early acid formation, the less time pyritic material is exposed to air, the less acid is formed. Thus, a positive preventative method is to cover pyritic materials as soon as possible with earth, which serves as an oxygen barrier. In mining, this is accomplished by current reclamation techniques and operating with only a small pit.

If the overburden also contains alkaline material such as limestone, acid water may not be discharged even though it is formed, because of in-place neutralization by the alkaline material. Discharges from this situation usually are high in sulfate.

Enough water to satisfy Equations 1, 2, and 4 usually is available in the overburden and coal material. Water also serves as the transport medium that removes oxidation products from the mining site and carries them into streams.

All techniques for preventing acid formation are based on control of oxygen. There are two mechanisms by which oxygen can be transported to pyrite--convective transport, and molecular diffusion [Ohio State University, 1971].

The major convection transport source is wind currents that can easily supply the oxygen requirement for pyrite oxidation at the spoil surface. In addition, wind currents against a steep slope provide sufficient pressure to drive oxygen deeper into the spoil mass. A factor to consider is the degree of slope after regrading. This is especially important on slopes subject to prevailing winds, since the wind pressure on the spoil surface increases as the slope increases. Thus, the depth of oxygen movement into the spoil would increase as the slope increases.

Molecular diffusion occurs whenever there is an oxygen concentration gradient between two points, such as between the spoil surface and some point within the spoil. Molecular diffusion is applicable to any fluid system, whether gaseous or liquid. Thus oxygen will move from the air near the surface of the spoil, where the concentration is higher, to the gas or liquid-filled pores within the spoil, where it is lower. The rate of oxygen transfer is strongly dependent on the fluid phases and is generally much higher in gases than in liquids. For example, the diffusion of oxygen through air is approximately 10,000 times greater than through water. Therefore, even a thin layer of water (several millimeters) serves as a good oxygen barrier.

The most positive method of preventing acid generation is by installing an oxygen barrier. Artificial barriers such as plastic films, bituminous, and concrete would be effective, but these have high original and maintenance costs and would be used only in special situations.

Surface sealants such as lime, gypsum, sodium silicate, and latex have been tried, but they, too, are high in cost, require repeated application, and have only marginal effectiveness. The two most effective barrier materials are soil, including non-acid spoil, and water. The minimum thickness of soil or non-acid spoil needed is a function of the soil's physical characteristics, soil compaction, moisture content, and vegetative cover. Deeper layers would be needed for a sandy, dry granular material with large grain size and porosity than would be required for a tightly packed, saturated clay that is essentially impermeable. Soil thickness should be designed on the basis of the worst situation--such as a dry soil where oxygen can move more readily through cracks and pore spaces devoid of water. A safety factor should be included to account for soil losses from such causes as erosion.

Vegetation not only serves to control erosion while it lives, but also becomes an oxygen-user through the decomposing process after it dies. This further aids the effectiveness of the barrier. The organic matter that is formed also aids in holding moisture in the soil.

Water is an extremely effective barrier when the pyritic material is permanently covered. Allowing the pyrite to pass through cycles where it is exposed to oxidation and then covered will worsen the AMD problem. Water barriers should be designed to account for water losses such as evaporation, and should include at least 30 centimeters (1 foot) of additional depth as a safety factor.

Additional measures to control AMD are water control and in-place neutralization. Water serves not only as the transport medium that carries the acid pollutants from the pyrite reaction sites, but it erodes soil and non-acid spoils to expose pyrite to oxidation. Facilities such as diversion ditches that prevent water from entering the mining area and/or carry the water quickly through the area can significantly reduce the amount of water available to transport the acid products. Sediment and erosion control are needed both during and following mining. Terraces, mulches, vegetation, and other measures used to reduce the erosive forces of water are effective in preventing further pyrite exposure. These measures usually are performed during reclamation.

Alkaline overburden material and agricultural limestone can be blended with "hot" acidic material to cause in-place neutralization of the acid and assist in establishing vegetation. In some cases grading directs acid seeps to drain through alkaline overburden. These techniques are more applicable to abandoned surface mines than to current mining, where proper overburden handling should prevent acid formation. The major exception may be those situations where an underground mine was breached and an acid discharge formed.

Acid seeps are a problem in certain locations. These may occur even though the surface mine has been properly graded and is supporting a good vegetative cover. Although the cause of acid seeps is not fully understood, it has been postulated that seeps occur when water percolating through the backfilled spoil reaches the impermeable underclay beneath the coal seam and is forced to move horizontally along the pit bottom and discharge at the toe of the backfill. In many cases the highly pyritic material associated with the coal is buried in the pit bottom, where it is subject to leaching by the horizontally flowing water. One solution to this problem is to place the acid-forming material above the pit floor and cover it with an impermeable material. A diversion ditch constructed along the top of the high wall keeps surface runoff from entering the fill area. Another solution would be to thoroughly mix the acid-forming material with an alkaline overburden material and then place it in the fill above the pit floor.

MINERAL POLLUTANTS

Fracturing the overburden material exposes numerous new surfaces to the oxidation and leaching process. Those ions subject to leaching are transported by water percolating through the spoil. Increases in concentration of calcium, magnesium, aluminium, sulfate, sodium, chloride, zinc, alkalinity, pH, and specific conductance have been reported [Curtis, 1972; Plass, unpublished manuscript; McWhorter et al., 1974]. The leaching process can be expected to continue for years, because under normal conditions the rate is very slow. In most mines, the rate of release is low enough that the increase in dissolved solids is not detrimental. Where extensive areas have been mined, however, the accumulation of the leachate may create problems. Little can be done to control this problem other than keeping the water that would cause leaching to a minimum, thus extending the process over a longer period.

ENVIRONMENTAL CONTROL OF SURFACE MINING

Grim and Hill [1974] have prepared a comprehensive report, Environmental Protection in Surface Mining of Coal, which describes methods of preplanning, mining, and reclaiming surface mines.

In summary, environmental control starts in the mine-planning stage. Data collected at the proposed mine site on the topography, overburden characteristics, ground and surface water, and climatic conditions are used to plan the mining technique to ensure (1) that proper erosion control is established, and (2) that selected overburden handling is accomplished to prevent acid mine drainage and ensure revegetation. Major irreversible problems have resulted when environmental concerns have not been considered along with extraction procedures. Good premining data collection and mine planning are the key to final recovery of the disturbed area.

Major advances in surface-mining methods have been made in the last few years. The collection of topsoil and subsoil for later spreading over the reclaimed spoil has been a major step in accelerating the recovery of stripped areas. Selective handling of the toxic overburden and its proper burial have reduced both acid and erosion problems. For mined areas on rolling to flat land, the grading of the land to a topography similar to that before mining has resulted not only in land more pleasing to the eye, but has made the land more useable and reduced the erosion threat because it is easier to revegetate. In mountain areas, the advent of mining methods with the colorful names of "head-of-hollow fill," "over the shoulder," "put and take," "block out," and "mountain-top removal" have led to ways of reducing the silt and landslides associated with steep-slope mining. Erosion-control practices are being adopted, but far more could be done in this area. Many states are now requiring sedimentation ponds below mining operations. While these are a major step forward in reducing the off-site sediment load, design modifications will be needed to make the ponds more effective.

Reclamation and revegetation know-how have increased significantly. The days of pine-tree plantings on six-foot centers are gone. The nutrient deficiency of the spoil and needs of vegetation are being taken into careful consideration. Lime, fertilizers, and mulches are being used. Better balanced mixtures of grasses and legumes are being planted. All-weather seeding is reducing erosion that occurred while awaiting the planting season. Nurse crops to hold the soil and assist permanent species are being utilized. Hydroseeding is more common in difficult areas. Good seedbed preparation is insuring better and faster establishment of vegetation. The practices lead to a quicker cover of a permanent vegetation which is the first step toward recovery.

Recovery of a surface-mined area has different meanings. Several states, by law, have defined recovery as a specific density of vegetation or cover. In other states, recovery means only that the mining operator attempted to revegetate the area in a "workmanlike manner." On the other end of the scale are those who believe that recovery has not been accomplished until the microflora and vegetation have returned to the premining condition. Recovery also can be measured as the return of water quality to premining conditions.

Several investigators [Curtis, 1972; Plass, unpublished manuscript; Hill, 1969] have reported that the dissolved ion concentration of water discharging from a disturbed area does not return to premining levels as soon as vegetation is established. Leacheable ions will continue to be discharged for years. The time required depends on the nature of the spoil material and climatic conditions. Spoils with high infiltration and percolation rates which are easily weathered and contain loosely bound ions will leach quickly. Leach rates will be higher in high precipitation areas. At the Elkins, West Virginia, mine reclamation project, sulfate, aluminum, iron, manganese, and acidity concentrations were still above background levels some seven years after a good vegetative cover had been established.

Smith, et al., [1974] recommend that spoil be considered as a young soil and treated accordingly. They recommend that spoil be classified under the U.S.D.A. soil classification system as *Spolents* under the order level of *Entisols*, which are defined as recent soils that have little or no evidence of development of pedogenic horizons. An intent here would be to bring to bear the knowledge of soils on the development of manageable soil units. Some investigations have studied the development of soil from spoil. Smith, Tryon, and Tyner [1971] studied 70- to 130-year-old spoils and reported that natural soil proved superior to old spoils in bulk densities, porosity, infiltration, nitrogen, and organic matter, whereas the old spoils were superior in depth of plant rooting, total available water-holding capacity, and certain plant nutrients. Others [Krause, 1964] have reported increased nutrient levels in spoils because nutrients had been leached and utilized by plants from the original soil, while certain strata in the overburden contained large quantities of potassium and other nutrients. Selective placement of the overburden material is critical to recovery. Acid, toxic, and physically poor material must not be placed near the surface. Material that is capable of supporting plant growth is required. The growth-supporting material may require additions of soil amendments such as lime and fertilizer. The most recent trend has been to remove "topsoil" and then replace it after grading. "Topsoil" here may be defined as natural soil capable of supporting plant growth. Several state laws now require top soiling and the proposed Federal Surface Mining Control and Reclamation Act of 1975 would require it for coal mining throughout the United States. Although top and subsoil may not always be ideal, they usually have certain advantages over other materials--including higher organic-matter content, better moisture-holding capacity, and the presence of natural seeds and soil microorganisms. Soil microorganisms are a key to the development of soil material, and fresh bedrock-type materials have insufficient organisms.

Recovery will be much faster and less fragile in the humid eastern United States than in arid and semi-arid western states. The low rainfall and saline soils of the West place additional stresses on recovery. Available plant species that will grow on these areas are limited, slow growing, and often difficult to establish. A popular belief in these states is that the acceptable cover should be native vegetation. However, the know-how on culturing native vegetation is limited, seed sources are scarce, and germination may be as low as 10 percent.

There are striking examples of surface-mined areas that have recovered to productive use as forest land, high-yield cultivated land, pasture land, recreational parks, and even home and industrial sites. On the other hand, there are many more examples of land that remains drastically disturbed. The keys to recovery at any site are attitudes and reclamation techniques of the mine operator, and the physical and climatic conditions at the mining site.

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Research Reports, Case Studies, Examples

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METHODOLOGY FOR ASSESSING RURAL NON-POINT SOURCE POLLUTION

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INTRODUCTION

Non-point source inputs have become the focal point of the Federal Water Quality Program since the recent finalization of comprehensive effluent guidelines and limitations for essentially all municipal, industrial, and agricultural waste point sources. The Environmental Protection Agency (EPA) has published two reports in compliance with Section 304 of Public Law 92-500 containing information for identifying and evaluating the nature and extent of non point sources [EPA, 1973a and 1973b]. These reports initially observe that agriculture, silviculture, construction, and mining activities contribute several pollutant substances to surface and groundwaters, and thus share with others the responsibility for protecting the quality of national water resources. Non-point sources generally fall into two distinct categories, rural and urban, and each has a set of unique characteristics. This paper addresses rural runoff.

Non-point agricultural pollutants are defined as organic and inorganic materials entering surface and groundwater from non-specific or unidentified sources in sufficient quantity to constitute a pollution problem. Included components are sediment, plant nutrients, animal waste, and pesticides from cropland, range land, pastures, and farm woodlots. Sediment is defined as the major pollutant and cropland the major source of sediment on a total mass basis, since 50 percent or more of the sediment deposited in streams and lakes is credited to agriculture [EPA, 1973 b]. However, a basic principle commonly overlooked is that "pollutants," inherent in natural land runoff, provide essential nutrients vital for productive aquatic eco-systems. It is mandatory, therefore, to establish natural or background levels and to assess the effects of increased inputs due to human activities on the aesthetic, recreational, and consumptive qualities of our surface waters.

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The EPA report entitled, "Methods for Identifying and Evaluating the Nature and Extent of Non-Point Sources of Pollution," [1973a] observes that

> "Planning for water quality management of watersheds and other planning regions can be effective only if the relationships involving input and output of various pollutants are understood, and if the effects of the pollutants on water quality can be reliably assessed. Further, it is essential that methods for predicting inputs and outputs be available to the planner."

The major thrust of our research is to develop and evaluate a survey design for field sampling techniques required to evaluate rural runoff, as well as to develop and verify models. One of the great difficulties of studies of this kind is determining the range of applicability of data and predictive relationships. An acre of cropland bordering a stream may contribute more pollution than an acre inland, and rule against the simple extrapolation of values derived from plot studies to larger areas. Detailed studies of a particular stream system may not produce findings applicable to other river basins, or upland data may differ from lowlands and wetlands. The assumption in this work has been that the greatest payoff for water-quality managers can be reached with a watershed-wide, statistically based, sub-basin type of study. This would tend to integrate the effects of variations in land use, soils, and topography.

THE CHOWAN RIVER BASIN PROJECT

The Chowan River basin of North Carolina and Virginia was selected for study because this basin has been under intensive investigation by the states of North Carolina and Virginia, EPA, U.S. Geological Survey, and the two state Water Resources Research Institutes. A great deal of data on point sources, stream chemical and biological dynamics, and effects on downstream water quality are being collected pursuant to the development of predictive models. The complementary benefits of conducting a non-point source study in the same basin utilized for point-source evaluation in the lower reaches motivated selection of this study area.

Scientific sampling survey methods are essential for this preliminary examination of non-point source inputs because of budget constraints. There are so many small drainage sub-basins within the Chowan River basin that only a very small fraction could ever be studied with the intensity required for reasonably complete knowledge. Even if the survey study is limited to relatively few parameters, only partial coverage is possible. Any inferences drawn from samples are incomplete knowledge and, therefore, must be supported by estimates of statistical precision.

Objectives

The specific objective and intrinsically compatible sub-objectives defined for this study are:

1. Develop and evaluate a survey design for field-sampling techniques required for rural runoff evaluation and model development and verification for small drainage sub-basins, with compatible emphasis on:

a. Cost effectiveness of various sampling precision levels, including comparison of grab techniques with instumental methods.

b. Modeling techniques for determining input from rural runoff and its effect on the chemical and biological characteristics of receiving streams.

c. Relationships between land use, physiography, rainfall-runoff, and the impact of these factors on receiving streams.

Project Scope

An interdepartmental group of researchers at North Carolina State University was assembled to formulate work plans necessary to achieve the multidisciplinary project objectives. D. H. Howells, director of the Water Resources Research Institute of The University of North Carolina, served as project manager. Co-investigators are F. J. Humenik, a sanitary engineer, and M. R. Overcash, a chemical engineer, in the Biological and Agricultural Engineering Department; D. W. Hayne, Departments of Statistics and Zoology; J. W. Gilliam, Soil Science Department; W. C. Galler, Civil Engineering Department; and A. M. Witherspoon, Department of Botany. Research assistants are F. Koehler, chemist; P. Geissler, statistician, and J. Preston, laboratory chemist. Cooperative efforts were established with the U.S. Geological Survey in North Carolina and the North Carolina Department of Natural and Economic Resources, and the Virginia State Water Control Board, and close liaison is being maintained with the complementary Chowan River Project.

SURVEY METHODOLOGY

Sampling Unit Selection

Site definition and selection were made by co-investigators from the Statistics Institute. Tactical considerations for sampling included the definition of what constitutes a basin small enough to be manageable for study purposes and big enough to be representative. Additionally, sampling routine, frequency, and intensity had to be determined. In the present instance, the primary sampling unit is a second-order stream, formed by the confluence of two first-order (no tributaries) streams. The definition of the sampling universe is based on USGS 1:250,000 scale maps.

First-order streams not included in the sampling frame also should be enumerated when the frame is drawn up. These will fall into several categories--including those with point sources, those where the relief pattern makes it clear that current will not be measurable, ephemeral streams, and the large number that flow directly into a higher-order stream and therefore do not conform to the special configuration specified for this study. By enumerating these, the proportion of all small basins included in the sampling frame can be calculated.

The definition between point source and diffuse inputs in these small rural drainage basins is extremely difficult. Subsurface flows from septic-tank fields in a rural community, flush water from a 10 to 15-dog kennel, or direct rainfall runoff from a 50-hog drylot operation have not been considered point sources. However, when dealing with streams where the base flow may be less than 0.1 cfs, the loads from these sources may very well obscure the effects of the diffuse inputs intended for study. The point-source definition dilemma for rural runoff finally was resolved by deciding to exclude only those inputs which currently have discharge permits; however, those streams where pseudo-point sources exist would be flagged for careful, individual analysis. The analysis will include these loads as part of the total rural watershed pollution load, but in land use/pollution potential considerations these anomalous situations would receive special attention.

Site Selection

Two major physiographic regions are included in the basin of the Chowan River--the Coastal Plain and the Piedmont. Within the Coastal Plain region, previous research has shown that water-quality parameters in drainage water from poorly drained soils differ from moderately well-drained or well-drained soils. The poorly drained soils have a higher content of soil organic matter, and water leaving these areas is higher in TOC. However, conditions within the poorly drained soils frequently favor denitrification, so less nitrate-nitrogen is transported from the poorly drained soils when compared with well-drained soils. Well-drained soils usually support a more intensive agricultural program and normally have greater slopes than poorly drained soils. Thus, the sediment losses from well-drained Coastal Plain soils would be expected to be greater. For these reasons, the Chowan River basin was stratified into three physiographic classes--poorly drained Coastal Plain, well-drained Coastal Plain, and Piedmont--to enhance the information gained in the study.

A sampling frame for each stratum was assembled and four sub-basins were chosen at random from each frame. Field trips to verify map and aerial photo observations resulted in disqualification of one site because an impounded lake obscured the upstream confluence point. This site was replaced by an additional randomly selected site. Characteristics of the study areas are as follows in Table 1:

TABLE 1

Characteristics of the Study Areas

Character of the Subarea	Designation	Approximate Area	Total Possible Sampling Units in Frame
Coastal Plain		693 sq mi	
Poorly drained	P		13
Well drained	W		11
Piedmont	M	253 sq mi	12

The confluence flow at all 12 sub-basins is grab-sampled to assess total nonpoint source load from the drainage watershed. From this total of 12 sub-basins, 4 basins (2 in the Coastal Plain and 2 in the Piedmont) were chosen for instrumented study. Additionally, 3 of the 12 sub-basins were chosen for mass balance investigations, as a check on internal consistency. Two sub-basins (one Coastal

Plain, one Piedmont) were chosen for kinetic and biological study. Various methods and precision levels of sampling are being used to evaluate cost-effective ratios. Data from the instrumented sites will be considered as field truth for precision comparison of various grab-sampling techniques conducted at all sites.

The designation and approximate areas of the selected sub-basins are as follows:

Poorly drained Coastal Plain:	P-11	6.5 sq. mi.
	P-9	5.2
	P-13	65.1
	P-10	7.8
Well drained Coastal Plain	W-3	8.3 sq. mi.
	W-8	5.0
	W-4	2.4
	W-10	7.5
Piedmont:	M-2	9.4 sq. mi.
	M-3	19.2
	M-1	11.6
	M-7	7.7

Sampling Schedule

The sampling schedule for uninstrumented sites was devised after designating three groups of four sub-basins each as a collective field trip. These groups roughly conform to the three physiographic regions. This restriction causes a loss in randomness, but was imperative due to time and travel-cost considerations.

After preliminary experimentation, a four-day sequence was established. Three days are devoted to data and sample collection in the field, and the fourth day is allocated to record and sample processing and equipment maintenance.

Based on this four-day sequence, a four-week sampling schedule evolved. This allows six field trips and one rest session per four-week period. The schedule is randomized on three levels: (1) group of sub-basins; (2) sub-basins, and (3) sampling points within sub-basin. The sampling plan for instrumented sites calls for weekly servicing of all four locations and is independent of and in addition to the grab-sampling schedule.

MEASUREMENTS

Grab Sites

Grab sampling was initiated on November 24, 1974. Prior to this time procedures were established, after which a field crew was trained. The overall procedure is separated into three steps: (1) make water-quality measurements; (2) obtain water-quality sample, and (3) gauge streamflow. The order is important, particularly in slow-flowing Coastal Plain streams, because dislodging bottom sediments by walking in the streambed would cause spurious results. The water-quality measurements are made from the bank on small streams or from a bridge on large streams. Parameters being measured in the field are temperature, dissolved oxygen, pH, conductivity, and streamflow.

Because streamflow is felt to be the least-precise measurement, attention is focused on obtaining the best estimates possible. Two stream types are present, Piedmont and Coastal Plain. Piedmont streams are amenable to measurement, but Coastal Plain streams presented a multiplicity of technical and practical difficulties.

Streamflow measurements are of four types: (1) current meter; (2) volumetric; (3) float, and (4) estimation. Current meter measurements, where possible, are made using standard USGS procedures [USGS, 1974a]. Where velocities are below detection limits, an attempt is made to increase velocity by restricting the stream channel. The three current meter models used are Price's, pygmy Price's, and Ott. The Price's meter is used during high-flow conditions. The pygmy Price's and Ott meters are used under low flow--i.e. low velocity and depth--conditions. The Ott meter has proven to give greater sensitivity than the pygmy Price's meter and is the method of choice for low velocities. When velocities are not measurable by a current meter, float measurements are made. Volumetric measurements are made at all stations where feasible and, if possible, are compared with meter results for precision checks because the volumetric measurement is felt to be the most precise method employed. When conditions preclude any of the above, estimation or indirect methods are employed, according to USGS recommendations [USGS, 1974b].

At the time a streamflow measure is taken, stage is recorded for development of a stage-discharge relation. This serves two functions. First, it provides an alternate, low-cost streamflow measuring technique. Second, it establishes an historical data base pursuant to translation of stage-recorder data to stream flow.

Mass Balance Sites: At the three mass-balance stations, each first-order stream is sampled approximately 20 feet upstream from the confluence and the main (second-order) stream is sampled approximately 20 feet downstream from the confluence. Comparison of the sum of the two first-order contributions with the second-order load is an additional estimate of precision.

Kinetic and Biological Sites: Each of the two basins serving as kinetic and biological study systems are sampled at six separate points. One first-order stream is sampled just before the confluence and approximately two miles upstream. The other first-order stream is sampled just before the confluence. The second-order stream is sampled just below the confluence, approximately one mile downstream from the confluence, and approximately two miles downstream from the confluence. At representative stream velocity of 0.3 ft/sec, the calculated time of travel for a 4-mile stretch would be approximately 19 hours. Questions still persist whether this stream reach allows sufficient time-distance for retrieval of meaningful kinetic data for both chemical and biological transformations.

Instrumented Sites: Site instrumentation consists of a streamflow stage recorder, recording rain gauge, rainfall collection apparatus, and an automatic water sampler. The water sampler is stage-activated and collects 28 discrete samples. These stations serve two purposes. First, since they are located at grab-sampling sites, a comparison of grab vs automated sampling can be made. Second, since the water samples are stage-activated, a determination of the correlation between streamflow and parameter concentration (sediment, NO_3, etc.) can be made.

Three of the four locations employ natural controls, i.e., the existing streambed. The fourth site, which will have a V-notch weir, is situated at a road crossing where the stream passes through a double-barreled concrete culvert. This again will allow a cost-precision comparison to be made between measurements before and after the weir installation.

DATA GENERATION

Analytical Techniques

Chemical: Chemical analyses have been run on all samples for TKN, o-PO_4, NO_3, Cl, TOC, and COD. Selected samples have been analyzed for t-PO_4, NH_3, and BOD. Nitrate and chloride have been analyzed routinely on a Technicon Autoanalyzer II, since sampling initiation. TKN, TOC, COD, t-PO_4 and BOD are analyzed using procedures given in Standard Methods. Suitable automated procedures for BOD and TOC are not available, and new technology for automated analyses involving digestion for TKN and t-PO_4 is under development.

Several short-term sub-studies of analytical techniques have been undertaken. An investigation of the possible applicability of a UV spectrophometric method for determining nitrate concentration indicates that organic material present in stream samples precludes use of this technique. Sample preservation studies have been investigated. Data analyses for acidification to $pH < 2$, cooling to 4°C, addition of 40 mg/l $HgCl_2$, and simple storage indicate that for a one-week period acidification and cooling are acceptable preservation methods. Results for nitrate [Figure 1], showed closed correlation between $pH < 2$ and 4°C preservation with initial unfixed values, while the unfixed sample varied with time. TKN values [Figure 2] for all preservation techniques are similar with initial sample value. The unfixed TKN value also varied with time. However, the sum of nitrate plus TKN is relatively constant, with time indicating total nitrogen conservation among metabolic forms. COD data [Figure 3] indicate that within the wide experimental variation characteristic of the procedure outlined in Standard Methods [APHA, et al. 1971], all preservation techniques are similar and a choice could not be made exclusively on the basis of this test. Thus, for grab samples, cooling from time of sample collection to analysis is employed and, for automated sampler samples, acidification is indicated.

In an attempt to increase sensitivity in TOC analyses, the possibility of sample size as a variable was investigated. Preliminary results indicate no dependence of sensitivity on sample size. To validate low-level TOC results, four samples were analyzed 10 times each. The mean varied from 12.5 to 23.0 mg/l. The range varied from 1.2 to 7.0 mg/l and σ varied from 1.4 to 4.3. Excluding one marginal value reduces the range of ranges to 1.5 and the range of σ's to 1.9. These findings greatly increase the amount of confidence placed on TOC analyses for these stream samples.

Various digestion times between 5 minutes and 2 hours have been investigated for the COD test, according to procedures outlined in Standard Methods, to determine the time required for total oxidation. The shorter digestion period allowed in Standard Methods would be routinely employed if it could be demonstrated that complete

FIGURE 1

Preservation Study Results for Nitrate (NO_3)

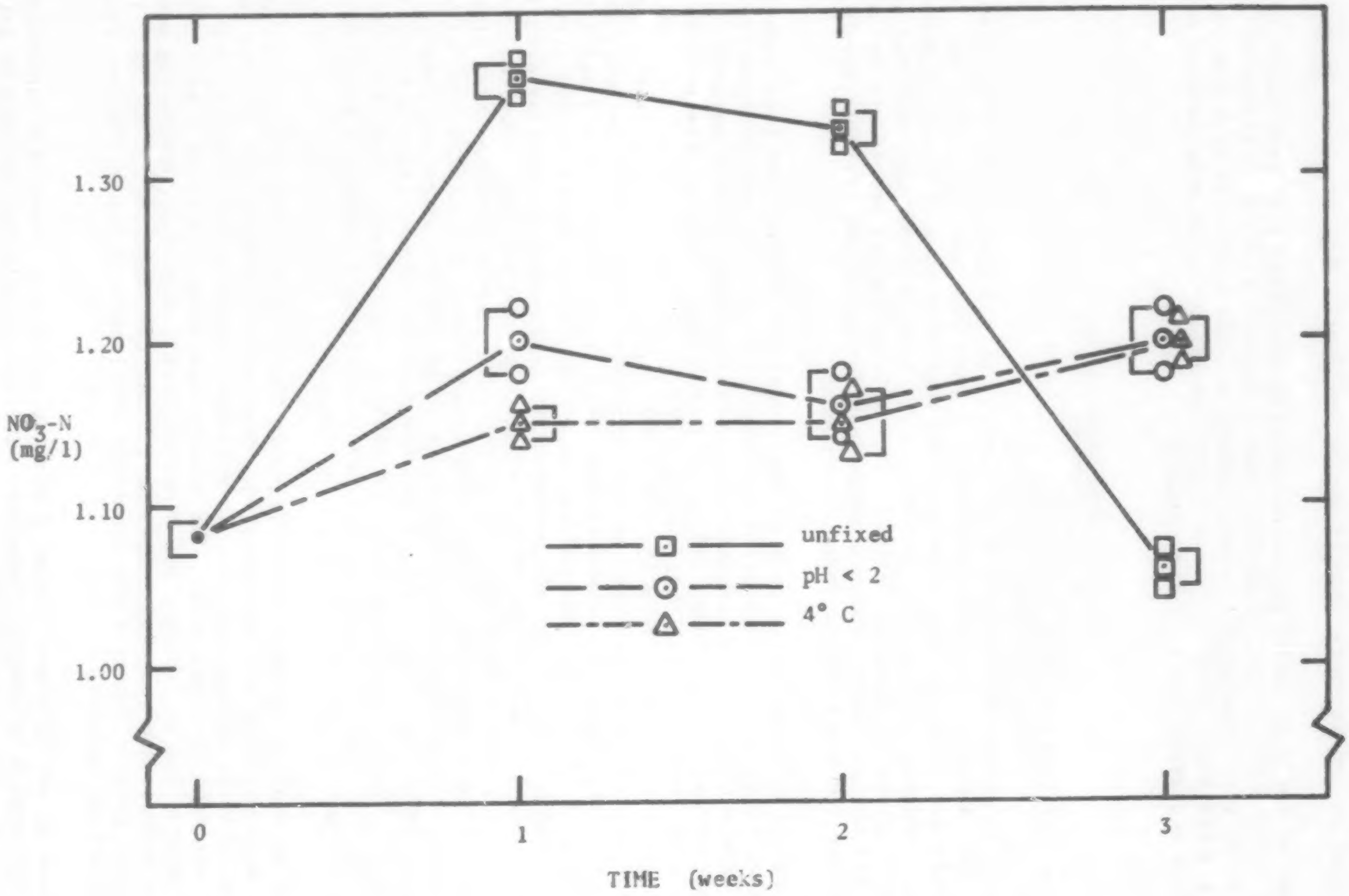


FIGURE 2

Preservation Study Results for Total Kjeldahl Nitrogen (TKN)

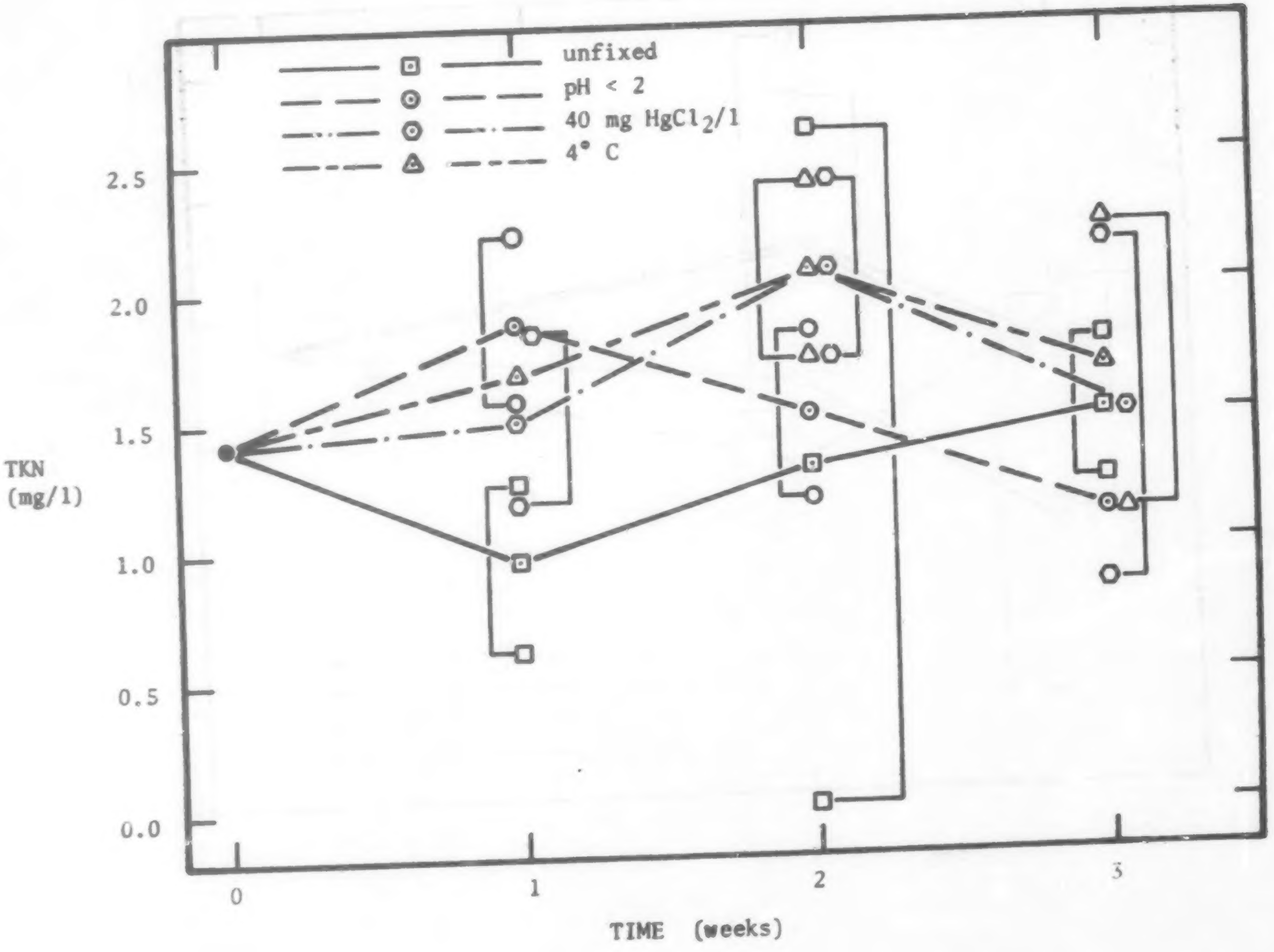


FIGURE 1

Preservation Study Results for Chemical Oxygen Demand (COD)

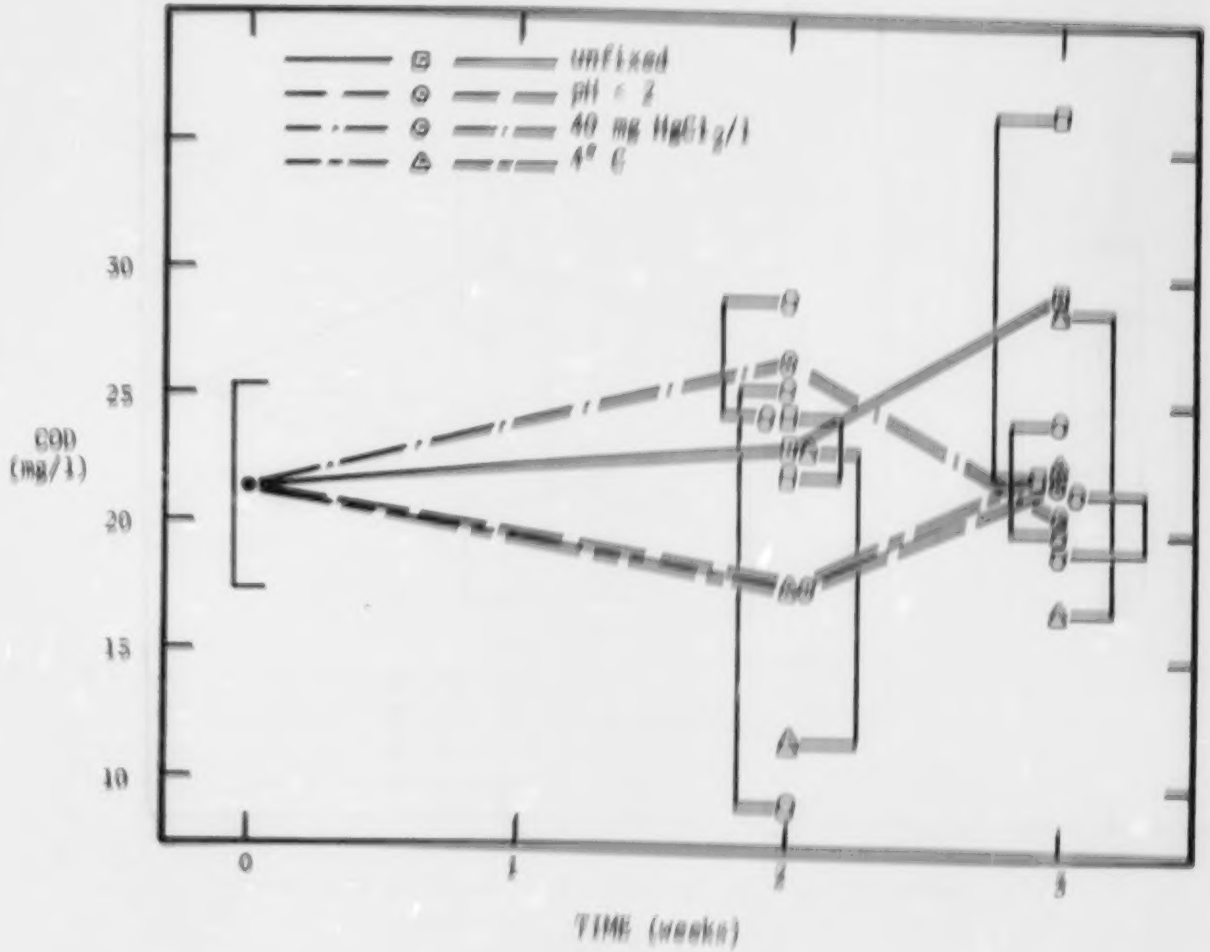
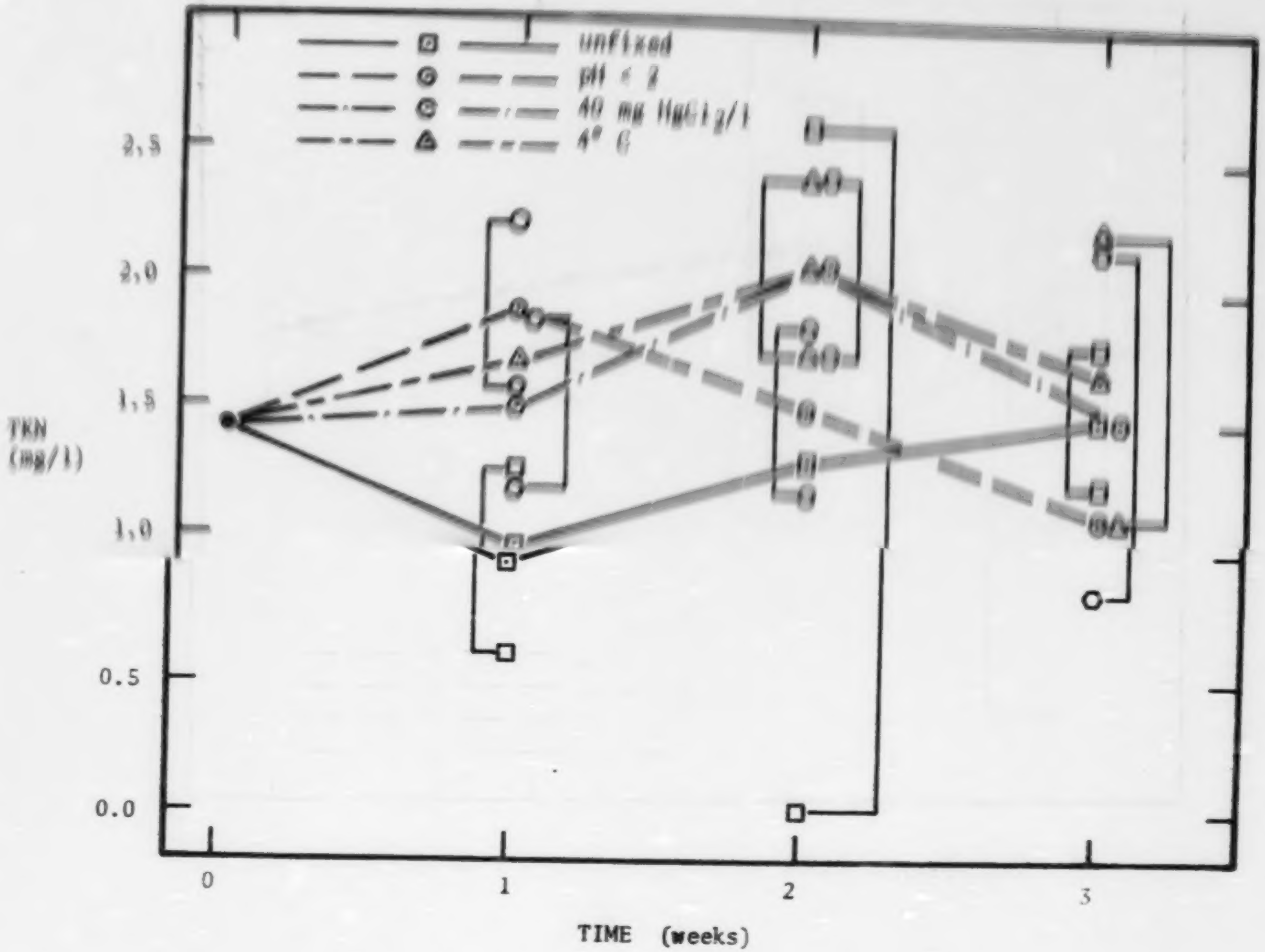


FIGURE 2

Preservation Study Results for Total Kjeldahl Nitrogen (TKN)



oxidation is achieved consistently in a certain fraction of the recommended two-hour digestion. Statistical analysis of results for these time boiling studies presented in Table 2 indicate that the total COD is realized in the first 30 minutes of digestion. Thus a 30-minute digestion period has been adopted for the COD analysis. Initial oxidation rates for smaller time increments during the first 30 minutes will be determined for use in developing BOD_5/COD relationships and rate constants. These conclusions remain tentative pending verification by more sensitive techniques, such as ultraviolet spectrophometric and potentiometric titration procedures currently being investigated to improve data precision.

During the remainder of the project, it is expected that all samples will be analyzed for TKN, t-PO_4, Cl, and NO_3 via automated procedures utilizing a Technicon Autoanalyzer II. To maintain the total workload within manageable proportions, the restriction will be made that approximately half of all samples will be analyzed for TOC, COD, and o-PO_4, and about a quarter of all samples will be analyzed for NH_3. Justification for the low intensity of NH_3 values will be established so estimates can be made of concentrations in unanalyzed samples.

Because sediment load and sediment-associated transport are important, half of all samples will be subjected to the following scheme: total concentrations of t-PO_4, TKN, COD, and TOC will be measured; the amount of non-filterable residue will be determined, and then the filtrate analyzed for the same chemical components. Thus will emerge an estimate of the sediment load in a stream (from non-filtraterable residue) and the amount of nutrient transport associated with sediment (the difference between total and filtrate concentrations).

Kinetic: Two stream systems of approximately 4 1/2 miles in length, one in the Piedmont and one in the Coastal Plains, were selected for intensive study to obtain biological and chemical kinetic data. (Detailed mapping and land-use evaluation of these areas are underway.) Six stations in each stream system are sampled every four weeks. Analyses of these samples yield five-day and long-term BOD evaluations over a 20 to 30-day period for both nitrogenous and carbonaceous materials at 10, 20, and 30 degrees centigrade as well as rate constants for oxygen uptake.

Laboratory BOD analysis and time-series analysis for NO_3, TKN, COD, and TOC on sample aliquots taken from the BOD bottles will be made for at least one total year seasonal cycle. BOD analyses are run on samples taken at other sites to determine rate constants for all sampling stations for comparison and additional data input for modeling.

Algal Assays: Algal assays are made on all samples which are analyzed for BOD to enhance the kinetic studies. Algal standing crops at stream stations designated for kinetic and biological study are determined, along with population dynamics. Data will be used to assess real and potential eutrophication levels and the suitability of algae as bio-indicators.

Land-Use Considerations: A cooperative effort has been undertaken with the Soil Conservation Service to determine the exact boundaries of each of the drainage sub-basins, and area measurements have been initated. Inital land classification types are forested, pasture, cultivated, livestock production, and housing. The information obtained from photographs will be supplemented by on-site visits. The land use within each site will be compared to the stream-quality parameters to discern if any relationship can be established on a watershed basis between land use

TABLE 2

Statistical Analysis of COD vs. Digestion Time

Sample		COD mg/l			
		15 min	30 min	60 min	120 min
1	$\bar{x}$	22	18	19	21
	s	4	6	7	4
	n	4	5	5	5
2	$\bar{x}$	10	12	19	16
	s	3	4	11	3
	n	4	5	5	5
3	$\bar{x}$	34	36	35	42
	s	3	4	3	3
	n	5	5	5	5

TABLE 3

Statistical Analysis of Water Quality Data for Different Physiographic Regions

Physiographic Region		Chemical Parameter mg/l			
		NO_3	TKN	o-PO_4	COD
P	$\bar{x}$	0.39	1.29	0.28	37.4
	s	0.31	0.82	0.52	16.6
	n	99	90	98	92
W	$\bar{x}$	0.54	1.13	0.039	26.5
	s	0.34	0.69	0.043	10.4
	n	35	32	36	33
M	$\bar{x}$	0.05	0.93	0.024	20.1
	s	0.068	0.64	0.060	9.9
	n	82	81	94	82

and receiving-stream quality during runoff events and base flow.

Modeling: The development of a mathematical model which has the capability to simulate the effects of rural runoff on receiving waters should be based on one governing precept: a model is a tool for use by planning and regulatory agencies. This simple idea invokes many criteria for a successful model. For instance, it must be user-oriented. An elegant model which requires sophisticated inputs and programming is not adapted to its audience. Again, a model not based on actual physical and chemical processes which describe effects resulting from causes is of little value in formulating regulatory guidelines. With this ideal in mind, an approach was formulated embodying three major themes: (1) attempt to establish predictive relationships for hydrology and pollutant transport; (2) attempt modeling in-stream transformations; and (3) incorporate simulation of pollutant effects on the aquatic ecosystem using algae as an indicator organism.

From the outset, it was realized that devising a model of this far-reaching magnitude was beyond the scope of the present project. It was decided to utilize existing models and models currently under development, modifying them where possible. As presently perceived, the modified Pesticide Transport and Runoff Model being developed by Hydrocomp, Inc., for assessing non-point source runoff will be utilized as a starting point. Data generated by the runoff model will be used as input data for the QUAL II instream water-quality model. Results of biological modeling being done by researchers under a concurrent lower Chowan River study will then be incorporated. After assembling the total model, an attempt will be made to calibrate and verify it, using the data now being collected. From this activity will emerge a mathematical model for simulating the effects on stream quality and aquatic environment resulting from existing or changed land-use activities.

DATA ANALYSES

Thus far, the emphasis in this study has necessarily been on getting underway and gathering data, with few detailed analyses performed to date. However, preliminary examination of the chemical data indicate trends. For example, data in Table 3 show that there is a large difference in NO_3 levels between the Piedmont and Coastal Plain streams. This large difference in means was anticipated, since the Piedmont basins are almost completely forested while there is extensive agricultural activity in the Coastal Plain basins.

If the variation between the poorly drained and well-drained areas is real, it may be due to denitrification in the anaerobic zones of the poorly drained soils. Examination of the COD data generally falls in line with land-use patterns following the expectation that organic loadings would be higher for cultivated areas than forested regions. However, the relatively high level of TKN in the Piedmont streams is surprising. An initial surmise is that it may be attributable to forest litter being either deposited in the stream or easily transported.

SUMMARY

A statistically based survey methodology for evaluating the impact of runoff from rural areas upon receiving streams has been developed and implemented on a

sub-basin level. Twelve sites equally distributed among the three characteristic physiographic regions in the total watershed are being sampled.

A major goal of this study is to determine the cost-effectiveness of various precision levels for obtaining data on a statistical basis to assess the impact of non-point sources. To accomplish this goal, the confluence low at all sites will be grab-sampled to evaluate the impact of runoff from the drainage area. Various methods and precision levels of grab-sampling will be investigated for cost-precision analyses. To represent the highest precision level possible, or "field truth," four instrumented sites containing equipment for continuous monitoring of rainfall and streamflow will be employed.

The large data base required for in-depth analyses and model development is currently being obtained. Two sub-basins in particular are being intensively studied for instream biochemical dynamics. Land-use parameters such as cropping patterns, management techniques, and human activities are being ascertained. This information will allow for model development, calibration, and verification.

The end result of these activities will be a comparison between survey and modeling methodologies as a management tool for land-use planning on a watershed basis, as evaluated by the ability to predict the impact of regulatory or management decisions on receiving stream quality.

ACKNOWLEDGMENT

This project was supported with supplemental funds from the Environmental Protection Agency Grant No. IBA023 SEERL, Pollution from Rural Land Runoff; the Water Resources Research Institute of the University of North Carolina, and the North Carolina Agricultural Experiment Station. Nat Thomas and Pat Walker, U.S. Geological Survey in North Carolina and Virginia, respectively, rendered vital assistance in site selection and stream-measurement techniques. Al Willett of the Virginia State Water Control Board assisted greatly, especially by lending gauging equipment.

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ANIMAL WASTE AND NON-POINT POLLUTION

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INTRODUCTION

Non-point agricultural pollutants are organic and inorganic materials entering surface and ground water from non-specific or unidentified sources. They include sediment, plant nutrients, pesticides, crop residues, and animal wastes from cropland, rangeland, pastures and farm woodlands. Sediment is the major pollutant in terms of weight and volume, and may be a significant carrier of pesticides, nutrients, organic materials, and pathogenic organisms.

At least a third of the pollutants entering United States waters are estimated to come from non-point sources. However, Congress did not address this problem until passage of the Water Pollution Control Act Amendments of 1972 (P. L. 92-500). A major goal of the Amendments is the concept of "zero discharge" of pollutants from non-point sources by 1985. This concept is somewhat ill-defined and, taken literally, may be technically, economically, and socially unfeasible.

This paper will briefly discuss non-point pollution resulting directly or indirectly from livestock and poultry production.

Current Effluent Guidelines and Limitations for the Feedlot Industry

Feedlot point-source effluent guidelines and standards were finally promulgated by the U. S. Environmental Protection Agency (EPA) in early 1974. A feedlot was defined to mean "a concentrated, confined animal or poultry growing operation for meat, milk or egg production, or stabling, in pens or houses wherein the animals or poultry are fed at the place of confinement and crop production is not sustained in the area of confinement [U. S. Environmental Protection Agency, 1974]. While feedlots as defined were interpreted to be point sources of pollution, experience indicates that most feedlots may also generate non-point pollution in the form of seepage, diffuse runoff, odors, gases, dust, insects, and other materials.

The EPA established a permit system with two standards of performance for pollution control for feedlot point sources. The degree of effluent reduction attainable by application of the best practicable control technology currently available (BPT) is defined to be no discharge of process wastewater pollutants to navigable waters, with runoff retention facilities to handle all

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process-generated wastewater plus runoff from a 10-year, 24-hour storm event for the particular point-source location. The BPT is to be implemented at existing feedlots by July 1, 1977. The degree of effluent reduction attainable by application of the best available technology economically achievable (BAT) is defined to be no discharge of process wastewater pollutants to navigable waters, with retention facilities to collect and hold all process-generated waste waters plus runoff from a 25-year, 24-hour storm event for the location of the particular source. The BAT is to be implemented for all new feedlot sources, and existing sources are to upgrade facilities to meet this standard by July 1, 1983. Waste pollutants will be permitted to overflow to navigable waters when rainfall, either chronic or catastrophic, cause an overflow from a retention facility designed, constructed, and operated to meet the applicable BPT or BAT standards.

For management reasons, EPA limited the eligibility requirements for permit application to feedlots as large or larger than 1,000 animal units (1,000 lbs. live weight approximately equals one animal unit). Recently, the U. S. District Court, District of Columbia, ruled that EPA cannot exempt producers with less than 1,000 animal units from the federal permit program. Unless the court decision is reversed on appeal, it appears that all operations that have a potential discharge, regardless of size, will be subject to the EPA permit program. From a practical standpoint, it may be expected that EPA will have to impose a priority system for permit issuance in the future.

Non-Point Contributions from Animal Wastes

Approximately 50 percent of the livestock and poultry in the United States are produced in confinement. These operations are normally considered point sources of pollution and are regulated by state and federal permit programs. The balance of U. S. livestock production is on pasture. In general, there is no practical method of treating process wastewater, polluted runoff, and accumulated manures from confinement operations. Consequently, most wastes from livestock production are returned to the land. Animal wastes applied to agricultural land come from three basic sources:

1. Wastes removed from feeding and confinement facilities.
2. Retained storm runoff and snowmelt from feedlots.
3. Excreta from livestock grazing on pasture.

Land application of animal wastes can increase pollution if proper disposal practices are not followed. When properly applied, the practice is highly effective and acceptable. Manure wastes can be beneficial in providing nutrients for growing crops and pasture, and can improve soil physical properties. This will aid in increasing infiltration, reducing runoff, and reducing soil erodibility.

Rates and methods of applying animal wastes on agricultural lands are diverse, depending on such factors as soil type, topography, climatic conditions, proximity to the ground water table and surface drainageways, and crops to be grown. A partial list of good management practices is as follows:

1. Estimate the plant nutrient value of the waste, and apply it on land uniformly in accordance with crop requirements. (The nitrogen requirement of the crop is often a limiting basis for determining the amount of waste to be applied.)

2. Schedule the time and frequency of manure applications for maximum nutrient utilization by plants. Capability for waste storage at the production site may be needed.

3. Obtain even distribution of wastes over disposal areas while avoiding contact with streams, grass waterways, and terraces.

4. If possible, incorporate manure into the soil as quickly as feasible following applications, or inject liquid wastes into the soil.

5. Ensure that enough land is available at the appropriate time for disposal of manure. For example, to maximize the utilization of waste, approximately 1 hectare (2.5 acres) of land will be required for every 3 to 6 dairy cows, 5 to 10 beef animals, 20 to 40 hogs, and 400 to 800 layers.

6. When large amounts of wastes are applied to the land, a highly productive crop should be planted to utilize the nutrients, reduce runoff, and reduce the amounts of nitrate and other pollutants that may reach the ground water.

7. Wet land, steep land, frozen or snow-covered land, and grassed waterways generally should not be treated with wastes since the material will not be readily absorbed and may result in polluted runoff.

8. Use good soil erosion-control practices to control runoff.

Normally, pasture production of livestock has not presented appreciable pollution problems except under special circumstances. An example of such a case is production on pastures adjacent to shellfish waters, where water-quality criteria may be much more critical than in swiftly flowing inland streams. Potential problems may also increase where animals tend to congregate around feeding, watering, and resting areas. Flowing streams also attract animals, especially during hot weather, and may increase pollution levels. These phenomena are not new. Early travelers through the American West frequently encountered difficulty in finding clean water; wild buffalo often lounged in rivers to drink, cool, and ward off insects. In Nebraska, the Pawnee Indians named one such river, "Buffalo Manure Creek" [U. S. Department of Agriculture, 1975].

Good management is the key to minimizing non-point pollution from pasture production of livestock. Relative importance of production practices will vary with classes of livestock and geographic regions. However, some commonly suggested, good management practices are:

1. Maintain an adequate land-to-livestock ratio. Avoid concentrations of animals that will create holding areas rather than grazing areas.

2. Maintain a highly productive forage on the land to retard runoff, entrap animal wastes, and utilize nutrients.

3. Plan a stocking density and rotation system of grazing to prevent overgrazing and eroding of the soil.

4. Locate feeders and waterers a reasonable distance from stream and watercourses. Move feeders to new locations often enough to avoid creating erodible paths by repeated livestock trampling.

5. Provide an adequate land absorption area downslope from feeding and watering sites, preferably with a filter strip of lush forage growth between such sites and receiving streams.

6. Provide limited access to streams and ponds. Use fencing to keep livestock from entering critical stream reaches.

7. Provide fences to prevent animals from wading in streams at points where they may concentrate for drinking. Fencing will usually be impractical, however, for many pasture operations.

8. Where the number of animals or the characteristics of land present critical pollution problems, run water from a stream farm pond, or well to watering troughs or tanks.

9. Provide summer shade, using trees or artificial shelters to lessen the need for animals to enter water for relief from heat. The same precautions used in locating feeders and waterers should be followed in locating shelters.

10. Install and maintain a good overall program of soil erosion control.

Research and Case Studies

Many early studies of agricultural pollution were oriented toward point-source problems and treatment of wastes. With passage of the Water Pollution Control Act Amendments of 1972, non-point sources of pollution were brought under constraints similar to point sources. A major problem in implementing the 1972 Amendments, particularly in the case of agriculture, lies in the difficulty of assessing the magnitude of non-point pollution ranging from a nationwide to the small watershed basis. Assuming that the magnitude of the problem can be assessed, there arises a second problem of developing and/or identifying preventive techniques and management tools which are applicable under a wide range of climatic, geological, agricultural, and environmental constraints for minimizing non-point pollution.

TABLE 1

Site Data and Yearly Mean Stream Pollution From Agricultural Land Runoff

Variable		Site F	E	K	P	X	Z
No. of animals*		0†	50‡	20§	42‖	38#	21**
Watershed area, ac		75	35	50	65	5	25
Soil types		sandy loams	sandy and fine sandy loams	loamy sands	fine sandy loams	sandy loams	sandy loams
Slopes, percent		0-25	0-15	0-6	2-10	2-10	0-25
Flow	cfs	0.109	0.040	0.047	0.055	0.006	0.020
	gal per day per ac	940	740	590	550	780	520
	gal per day per animal	-	510	1540	850	100	620
Study period		12/68-8/69	12/68-8/69	4/68-8/69	6/69-4/70	2/70-4/70	11/69-4/70
No. of samples		148	176	100	108	27	78
FC	10^3 col per 100 ml	10.0	190	370	9.6	0.2	30.7
	10^9 col per day per ac	0.33	5.12	8.2	0.19	0.01	0.58
	10^9 col per day per animal	-	3.58	20.4	0.30	0	0.69
BOD_5	mg per l	2.0	4.7	6.4	5.2	265	9.8
	lb per day per ac	0.015	0.028	0.031	0.023	1.66	0.041
	lb per day per animal	-	0.020	0.079	0.036	0.218	0.049
TOC	mg per l	14.6	9.6	17.6	14.2	322	32.5
	lb per day per ac	0.111	0.057	0.087	0.063	2.02	0.136
	lb per day per animal	-	0.040	0.216	0.097	0.266	0.162
N	mg per l	1.4	3.7	1.7	3.5	62	2.6
	lb per day per ac	0.011	0.022	0.008	0.015	0.389	0.011
	lb per day per animal	-	0.015	0.021	0.024	0.051	0.012
PO_4	mg per l	0.2	1.2	1.9	1.2	-	1.1
	lb per day per ac	0.002	0.007	0.009	0.005	-	0.005
	lb per day per animal	-	0.005	0.023	0.008	-	0.006

*1000 lb live weight basis

†Free of domestic animal wastes

‡200 sows on 3 acres of dry-lots plus wastes from 300 confined hogs spread on 5 acres

§200 hogs on 6 acres of dry-lots

‖60 tons of poultry wastes plus shavings spread on 15 acres, yearly

#22 tons of poultry wastes spread on 4 acres, once

**35 beef animals on 15 acres of pasture

From J. W. D. Robbins, et al. in Quality of Effluent from Farm Animal Production Sites. Proceedings of the International Symposium on Livestock Wastes, Columbus, Ohio. 1971.

TABLE 2

Precipitation Characteristics

Constituent	Concentration under Given Conditions* (mg/1)							
	1963-64 Urban[1]	1963-64 Rural[1]	Cooper[2]	Northern Europe[3]	1963-69 Forest[18,22]	Feth[5]	4 yr Ohio[4]	oyner[1]
Nitrogen								
NH_4-N	--	--	--	0.06	0 16	0.17-1.5	1.1	--
NO_2-N	--	--	0.14	0.31	0.30	0.56	1.15	--
Inorganic N+	0.7	0.9	--	--	--	--	--	--
Total N	1.27	1.17	--	--	--	--	--	0.73
Phosphorus								
Total PO_4-P	--	--	--	--	0.008	--	0.02	0.04
Hydrolyzable PO_4-P	0.24	0.08	--	--	--	--	--	--
Suspended solids	13	11.7	--	--	--	--	--	--
COD	16	9	--	--	--	--	--	--
Major ions								
Ca	--	--	0.65	--	0.21	--	--	--
Cl	--	--	0.57	--	0.42	--	--	--
Na	--	--	0.56	--	0.12	--	--	--
K	--	--	0.11	--	0.19	--	--	--
Mg	--	--	0.14	--	0.16	--	--	--
SO_4	--	--	2.18	--	3.1	--	--	--
HCO_1					0			

* Data are primarily yearly averages; numbers in headings refer to references.
+ Inorganic N = NH_4, NO_2, and NO_3-N.

From R. C. Loehr in Characteristics and Comparative Magnitude of Non-Point Sources. Journal Water Pollution Control Rederation, August 1974.

Many studies are currently proposed or underway which address the problem of non-point pollution from animal wastes. The following discussion involves several studies which show representative findings of the magnitude of animal wastes as non-point sources of pollution.

Robbins, et al. [1971, 1972] summarized findings of a two-year study to determine the importance of animal wastes in agricultural runoff. Their study demonstrated the relative pollutional effects of manure disposal and pastured animals [Table 1]. Major conclusions relative to non-point sources were:

1. **Natural pollution load on streams draining agricultural basins free of farm animals can be appreciable under certain conditions of rainfall and runoff.**

2. **Soil provides excellent natural treatment of animal wastes, and the extent of water pollution from animal pro-**

TABLE 3

Reported Precipitation Characteristics--Average Nitrogen and Phosphorus

Location	N		P	Reference
	Total* (kg/yr/ha)	NO_3+ NH_4-N (kg/yr/ha)	(kg/yr/ha)	
World mean	6.2 (0.8-7.0)	--	--	11 (pre-1954)
World mean	8.7 (1.8-22.2)	--	--	11 (1905)
Europe and U. S.	--	0.8-2.1	--	11 (pre-1952)
Temperate zone	--	6.8	--	11 (1938)
Humid temperature zone	5.6	--	--	11 (1960)
New York	10.0	--	--	11 (pre-1948)
United Kingdom				
Upland	8.2	--	0.27	12
Northern	8.7-19	--	0.2-1.0	12
Netherlands				
Rural	8.5	--	--	13
Industrial	16-100	--	--	13
Canada				
Hamilton, Ont.	--	6.0	--	14 (1948)
Ottawa	7.7 (4.8-12.9)	--	--	14 (1924-25)
Ceylon	--	12.9	--	14 (1941)
Western Australia	0.5-3.2	--	--	3
Scotland	8.2	--	0.45-0.7	10 (1962-63)
Ithaca, N. Y.	--	7.4	0.05	15
Aurora, N. Y.	--	7.6	0.06	15
Geneva, N. Y.	--	8.3	0.05	15
Hubbard Brook, N. H.	--	5.8	0.10	16
Cincinnati, Ohio+	9.6	5.2	0.6	9

* Data in parentheses indicate range of data.
+ Average U. S. rainfall of 30 in. (76 cm)/yr assumed.

From R. C. Loehr in Characteristics and Comparative Magnitude of Non-Point Sources. Journal Water Pollution Control Federation, August 1974.

TABLE 4

Characteristics of Runoff from Pasture Land and Land Used for Manure Disposal

Location and Conditions	Constituent						Remarks	Reference
	BOD	COD	TOC	NO_3-N	Total N	Total P		
North Carolina (g/day/ha)	30	--	63	--	7.6	2.8	Mixed grains and orchard, swine waste spread	40
	17	--	124	1.8	10.4	0.6	Pasture, corn, orchard, swine waste spread	
	35	--	97	1.7	7.7	3.5	250-hog drylot, row crops, wood, grassland	
	94	750	150	4.0	38	25	Pasture for 50-100 dairy cows, plus corn	
	3,450	10,700	3,750	46.5	400	130	Pasture for 160 cows on 15 acres	
	26	320	72	8	10	1.8	Poultry waste spread on 5 acres 3 times/yr	
	1,850	4,350	2,250	14	435	--	22 tons of poultry waste spread on 4 acres once	
	46	720	150	--	5.8	1.8	35 beef cows on 15 acres of pasture	
Wisconsin (kg/yr/ha)								
Manure not spread	--	--	--	--	4.4 (3.6-5.5)	1.3 (1.2-1.5)	3-yr average--1967-69, dairy cattle manure spread at the rate of 15 tons/acre	41
Manure applied*								
Winter	--	--	--	--	12.7 (3.0-27)	2.9 (1.0-5.8)		
Spring	--	--	--	--	3.8 (3.0-5.2)	0.8 (0.7-1.0)		

* High values due to a thaw and a 0.75-in. rain immediately after spreading manure in winter; manure spread in the spring was incorporated into the soil after spreading.

Note: Acre x 0.405 = ha; in. x 2.54 = cm.

From R. C. Loehr in Characteristics and Comparative Magnitude of Non-Point Sources. Journal Water Pollution Control Federation, August 1974.

duction is more dependent upon waste management methods than upon the total volume of waste generated.

3. Animal wastes should not be deposited directly in streams or waterways.

4. Points of animal concentration should be located away from streams and away from steep hillsides or depressions draining directly into streams.

5. Interceptor areas of vegetation should be provided between concentrated animal areas and drainage paths or surface waters.

6. Pollution indices for streams draining land receiving manure paralleled stream hydrographs except for slower recession than surface runoff.

Following good manure disposal techniques or with animals on pasture, the average BOD_5 loads in streams were less than 2 percent of the BOD_5 of animal wastes deposited in the watersheds. Even with poor practices of depositing high rates of manure on bare soil and in defined drainage paths, stream loads were less than 10 percent of the defecated raw waste load, which is not inconsistent with other reported research.

Loehr [1974] summarized pertinent literature and commented on feasibility of controlling certain non-point sources entering surface waters. He cautioned that the ecological impact of non-point sources depends on intensity and yield factors as well as on the seasonal distribution of their contribution. Because of the flow-dependent, intermittent nature of sources, a comparison of non-point sources based solely on concentration units is difficult and inaccurate. A better approach would be to use area yield rates such as quantity of characteristic per unit of drainage area per unit of runoff. Data on pollutant concentrations in precipitation were compiled and are reproduced here as Tables 2 and 3. Similarly, characteristics of runoff from pasture land and land used for manure disposal in two different geographical and climatical regions were presented and are shown here as Table 4.

Loehr compared and summarized pollution data from non-point sources [Table 5], noting that reported values should not necessarily be interpreted as amounts of materials necessarily reaching surface waters. Many of these parameters can be altered during transport across or through soil and during transport in creeks and streams. Figures 1 and 2 indicate relative ranges of total nitrogen and total phosphorous from all sources discussed. Yield of total nitrogen from land receiving manure under proper management was no higher than for crop and forest land receiving no manure. Maximum total phosphorous yield for land receiving manure and crop land was also similar. The range of values of total nitrogen and phosphorous was not as broad for land receiving manure as for crop and forest land. Also of interest is that yield rates were similar for precipitation, land receiving manure, and other land uses, while much higher yields resulted from manure seepage and animal feedlot runoff. The latter sources would normally be considered as point sources of pollution and would require more definitive control measures.

TABLE 5

Summary of Non-Point Source Characteristics*

Source	Concentration (mg/l)				
	COD	BOD	NO_2-N	Total N	Total P
Precipitation	9-16	12-13	0.14-1.1	1.2-1.3	0.02-0.04
Forested land	--	--	0.1-1.3	0.3-1.8	0.01-0.11
Range land	--	--	--	--	--
Agricultural crop land	80	7	0.4	9	0.02-1.7
Land receiving manure	--	--	--	--	--
Irrigation tile drainage, western U. S.					
Surface flow	--	--	0.4-1.5	0.6-2.2	0.2 -0.4
Subsurface drainage	--	--	1.8 -19	2.1 -19	0.1 -0.3
Cropland tile drainage	--	--	--	10-25	0.02-0.7
Urban land drainage	85-110	12-160	--	3	0.2 -1.1
Seepage from stacked manure	25,900-31,500	10,300-13,800	--	1,800-2,350	190-280
Feedlot runoff	3,100-41,000	1,000-11,000	10-23	920-2,100	290-360

* Data do not reflect the extreme ranges caused by improper waste management or extreme storm conditions; the data represent the range of average values reported to previous tables.

From R. C. Loehr in Characteristics and Comparative Magnitude of Non-Point Sources. Journal Water Pollution Federation, August 1974.

Table 5 (continued)

Area Yield Rate (kg/yr/ha)					Surface Area of Interest
COD	BOD	NO_2-N	Total N	Total P	
124	--	1.5-4.1	5.6-10	0.05-0.06	Total land area
--	--	0.7-8.8	3-13	0.03-0.9	Forest area
--	--	0.7	--	0.08	Range land
--	--	--	0.1-13	0.06-2.9	Active crop land
--	--	--	4-13	0.8 -2.9	Crop or unused land used for manure disposal
--	--	--	3-27	1.0 -4.4	Irrigated western soils
--	--	83	42-186	3-10	Irrigated western soils
--	--	--	0.3-13	0.01-0.3	Active crop land requiring drainage
220-310	30-50	--	7-9	1.1-5.6	Urban land areas
--	--	--	3	--	Manure holding area
7,200	1,560	--	100,1,600	10-620	Confined, unenclosed animal holding areas

CONCLUSION

The degree of control required for non-point pollution from animal wastes should be based on the relative importance of the various non-point sources in specific locations and on what is technically and logically controllable. In many cases it appears that it may be difficult to distinguish between pollution from farm animal wastes and natural, or "background," levels of pollution. So, even if significant effort were expended to absolutely eliminate pollution from farm animal wastes, little or no change may be noted in stream quality. In many cases it can be argued as to whether or not all non-point sources can be controlled or eliminated. If collection and treatment is anticipated, it should be remembered that the costs associated with conventional waste-treatment systems are prohibitive, especially when compared with the economics of land disposal and nutrient utilization of animal wastes.

FIGURE 1

Comparison of non-point sources,
giving range of total N and P concentrations.

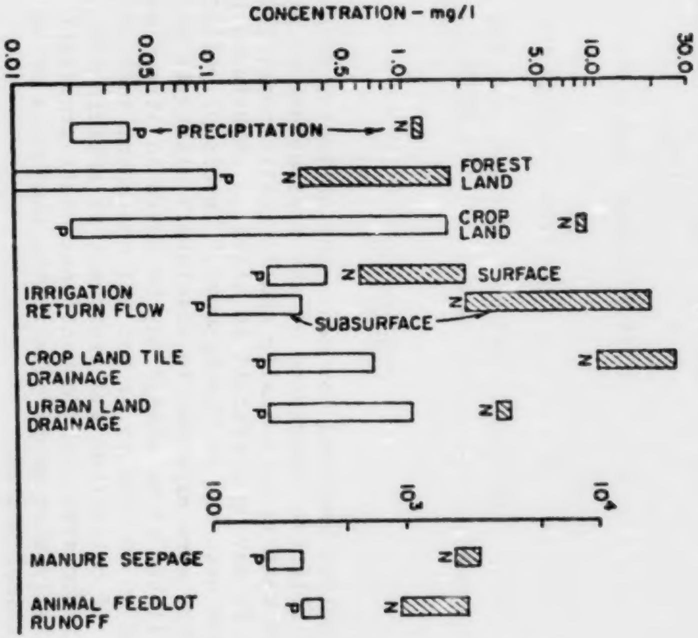
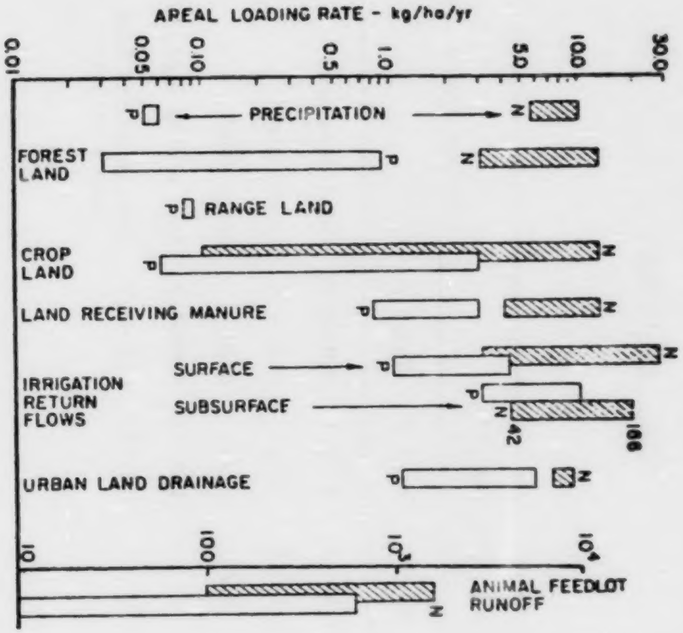


FIGURE 2

Contributions of total N and P
by various non-point sources.



From R. C. Loehr in Characteristics and Comparative Magnitude of Non-Point Sources. Journal Water Pollution Control Federation, August 1974.

A number of problems exist in using research information to develop sound management and regulatory approaches. A multitude of methods is available for evaluating the various pollution parameters. Many standard analytical tests used to characterize waters and wastewaters are not reliable for measuring pollution from animal wastes. Hence, it may be impossible to make reliable comparisons of different research data.

If non-point source criteria which are developed relate to stream water-quality criteria, compliance for a majority of livestock producers will be relatively feasible. In most cases all that will be entailed will be the practice of good soil conservation measures which have been observed for many years, and applying wastes at levels where they can be assimilated and utilized. Research to the present tends to support this approach as being the most prudent course to reduce non-point pollution from animal wastes.

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AGRICULTURAL CHEMICALS AND THEIR IMPORTANCE AS A NON-POINT SOURCE OF WATER POLLUTION

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INTRODUCTION

The use of agricultural chemicals in the production of food and fiber has been increasing steadily over the past several decades. Annual nitrogen, phosphorus, and potassium fertilizer use in the United States now amounts to approximately 4.3, 3.4, and 2.7 million tons, respectively [Smith, 1967]. Production and use of herbicides, insecticides, acaracides, fungicides, and nematocides now amounts to over 1 billion pounds per year [U. S. Tariff Commission, 1967]. The threat of contamination of our environment by these materials is of concern. It was the basis for the publication of several popular books, including Silent Spring [Carson, 1962], Pesticides and the Living Landscape [Rudd, 1964], and Since Silent Spring [Graham, 1970]. Several books also appeared which defended the use of organic pesticides including The DDT Myth [Beatty, 1973] and That We May Live [Whitten, 1966]. The threat from agrichemical pollution has been the subject of numerous scientific books [Gould, 1966; Brady, 1967; Miller, 1969; Ameri- Chemical Society, 1969; Gould, 1972, Gunzi, 1974] and governmental reports [Whitten, 1966; U.S. Department of Agriculture, 1966; U.S. Department of Health, Education, and Welfare, 1969; U.S. Department of Agriculture, 1970; Council of Environmental Quality, 1973; Pimentel, 1971; Evens and Duseja, 1973]. The detrimental effects of the "hard" insecticides and pollution resulting from fertilizers and animal wastes on fish and wildlife have been expounded upon in detail in these reports.

The objectives of this paper are to: (1) briefly review the findings of these reports; (2) summarize the potential hazards to our environment resulting from chemicals which are presently being used; (3) provide an index of the potential hazards of agricultural chemicals which are used in normal crop production systems and their relative importance, and (4) recommend ways to minimize these hazards if they exist.

A REVIEW OF AGRICULTURAL CHEMICAL POLLUTION

"Hard" Insecticides

Nearly all of the detrimental effects to fish and other wildlife which have been reported can be attributed to what might be called the "dirty dozen." These, the chlorinated hydrocarbon insecticides, include aldrin, BHC, chlordane, DDE, DDT, dieldrin, endrin, heptachlor, lindane, methoxychlor, TDE, and toxaphene. Four major factors were responsible for the detrimental environmental effects which resulted from the use of these chemicals. First, they were very persistent, most having half-lives greater than 5 years [Edwards, 1970]. Hence, the name "hard" pesticides. Second, they were extremely toxic to fish, the majority having LC_{50} values of 9 ppb or less to rainbow trout and bluegill. Third, the chemicals were widely and indiscriminately used more as a preventative measure than as a cure in controlling insects. Last, some of the chemicals were found to biomagnify in aquatic organisms and to have detrimental effects on the reproduction of certain birds [Edwards, 1970].

In surveys of the nation's waters, small amounts of all the chlorinated hydrocarbon insecticides were detected in many of the samples [U. S. Department of Agriculture, 1966; U.S. Department of Health, Education and Welfare, 1969; Green, et al., 1967]. Concentrations ranged from nearly insignificant to 17 ppb. In soil samples, concentrations generally ranged from nearly insignificant to 2.2 ppm, but in a few orchard soils DDT levels as high as 287 ppm were found [U.S. Department of Agriculture, 1966; Sheets, 1967].

DDT was shown to biomagnify in aquatic organisms to levels as high as 1600 ppm [Hunt and Bischoff, 1960] and to have detrimental effects on the hatchability of eggs from certain bird species [U.S. Department of Health, Education, and Welfare, 1969; Edwards, 1970].

"Permanent" Pesticides

Another group of agricultural chemicals that were reported to be potential hazards to the environment were the "permanent" pesticides [U.S. Department of Agriculture, 1966]. These included chemicals that contained metallic elements such as mercury (seed-treatment fungicides), arsenic (insecticides, herbicides), manganese (fungicides), and zinc (fungicides). Since the elements themselves are not further degraded, the chemicals have been termed permanent. Arsenic, lead, and copper accumulations in some orchard soils were so high as to become toxic to trees [Jones and Hatch, 1945] and tree seedlings [Reuther and Smith, 1953]. Lead and mercury-bearing pesticides are not used in general crop production, while tin, copper, manganese, and zinc compounds are used at very low rates of application. Because some of these chemicals do biomagnify in plant and animal life, they are thought to constitute a potential hazard to the environment [Lagerwerff, 1967]. However, all of these metallic pesticides are readily complexed to soil colloids and are relatively immobile, so their threat to water pollution is relatively low.

Another group of agricultural chemicals that may be included in the "permanent" pesticides group are the cationic herbicides diquat and paraquat. These chemicals are irreversibly "fixed" on the internal surfaces of certain types of soil clay minerals and hence are not available for biological degradation [Weber and Scott, 1966; Weber and Coble, 1968; Weber, et al., 1969]. They are ionically bound to soil colloids and are immobile, hence they are not potential water pollutants.

Fertilizers

Fertilizers also have been reported as sources of water pollution. They contribute to algal blooms and other forms of eutrophication [Verduin, 1967]. However, it is estimated that 1 percent of the phosphorus fertilizer applied to soils of the United States ends up in runoff and drainage waters. This is a very small amount, relatively speaking, especially in comparison to the amount of phosphorus pollution resulting from municipal sewage and phosphorus-containing detergents. Phosphorus applied to soils in the form of agricultural fertilization is readily complexed to soil colloids and becomes relatively immobile, hence it does not present a very significant threat as a water pollutant, except when the soil particles themselves move.

Potassium added to soils in agricultural fertilizers also does not present a potential water-pollution hazard,as the element is readily adsorbed to the exchange sites on soil colloids.

The only fertilizer nutrient which was reported to be a potential water pollutant was nitrogen [Smith, 1967]. Nitrogen applied in the ammoniacal [NH_3 gas] or ammonium [NH_4^+] ion form does not pose a water-pollution hazard, as both forms are readily adsorbed by the soil colloid exchange sites and are hence immobile. Nitrate [NO_3^-] nitrogen, however, may present a problem, as this form of nitrogen is very mobile in soils. It dissolves in runoff waters and leaches readily in soil. Correlation studies between the amount of nitrogen fertilizer applied and the nitrate content of water samples, however, showed no relationship [Smith, 1967]. Surface-water samples taken from rural water supplies did contain from 0.5 to 0.75 ppm nitrate nitrogen, but there was little correlation between nitrates in the water and algae growth.

Soil Erosion

Probably the greatest contributor to water pollution is soil erosion. Soil particles not only physically impair streamflow and cause flooding and navigation problems and fish kills, but the particles have adsorbed on their surfaces the nutrient elements and pesticides which may be released to cause detrimental effects to aquatic life. It is estimated that approximately 3.5 billion tons of soil are eroded from privately owned land in the United States each year [Council on Environmental Quality, 1973]. This soil is carried in the approximately 1.16×10^6 million gallons of water which run off our lands each day [U.S. Department of Health, Education, and Welfare, 1969]. Much of this erosion occurs as a result of urban development. Renewed interest in the use of soil-conservation practices are helping to reduce soil erosion and runoff, but many of our rivers are still dirty, unsafe, and unusable, and flooding remains an annual disaster.

POTENTIAL WATER POLLUTION HAZARDS OF CHEMICALS USED TODAY

Most of the "hard" insecticides are no longer used in general crop production, and the few that remain are used under very restricted conditions. Because of the persistence of these chemicals in the soil, they will continue to be found at very low concentrations in the environment for many years. However, their importance as sources of water pollution will continue to diminish. Many of the permanent pesticides containing metallic elements (Hg, Pb, Sn, Mn, Cu, Zn) are applied in only very small amounts or are so restricted in their use as to no longer present a potential hazard to water pollution. Others, such as the arsenic-containing and cationic herbicides, although of a permanent nature, are readily fixed in the soil and do not pose a significant threat at this time. With the introduction of several new cationic herbicides and the continued use of arsenic herbicides, the accumulative effects of these chemicals in the soil should continue to be monitored.

Herbicide runoff studies carried on in the southern United States showed that fluometuron--a nonionic, phenylurea herbicide with a water solubility of 90 ppm--was relatively mobile, but its rate of degradation was faster than its rate of leachability [Peeper and Weber, 1974; Savage, 1975; Wiese, 1975]. Only small amounts (less than 1 ppm) of the herbicide were found in runoff waters and the chemical did not persist from year to year. Studies of a similar nature are needed on pesticides of higher mobility and longevity than fluometuron.

Recent studies [Gambrell, et al., 1973] concerning the loss of nitrogen fertilizer from soils showed that significant amounts of nitrate-nitrogen percolated through the soil and ended up in the tile drains that carried them to streams and waterways. With increasing amounts of nitrogen fertilizers being applied to the land each year, the potential hazards of water pollution from nitrates deserves to be examined further.

The replacement of the cholorinated hydrocarbon insecticides by organophosphorus, carbamate, and other relatively short-lived chemicals will greatly decrease the potential water-pollution hazard from insecticides. Although the phosphorus-containing insecticides are considerably more toxic to man, they have important features that make them much more environmentally safe. These features include: relative immobility in soils, short half-lives, and very low biomagnification potentials in plants and animals.

Most of the herbicides, with the exception of the cationic ones and the arsenates, are very safe, environmentally. The greatest safety features of herbicides are that they are generally not very toxic to fish and do not biomagnify. Their mobility and longevity varies greatly.

AN INDEX OF AGRICULTURAL CHEMICAL POLLUTION POTENTIAL

Various processes and factors are involved in the dissipation of agricultural chemicals in the environment. These include degradation processes such as chemical, biological, and photochemical decomposition, and transfer processes such as absorption, exudation, retention and accumulation by organisms, adsorption by soil colloids and other surfaces, and movement in the vapor, liquid, and solid state through the atmosphere, soil and water [Figure 1].

These degradation and transfer processes are dependent upon edaphic factors such as the types and amounts of soil colloids present, soil pH, soil moisture and temperature, types and numbers of soil organisms, and soil nutrient level, and climatic factors such as temperature, humidity, rainfall intensity, duration, and occurrence, and radiation quality and intensity. These processes are discussed in detail elsewhere [Weber, 1973; Weber, 1972a; Weber, 1972a], but in essence whether or not a chemical will be a potential water pollutant depends upon four key factors. These are: (1) mobility; (2) longevity; (3) toxicity, and (4) biomagnification.

I have proposed that these four factors be used in an evaluation scheme for

FIGURE 1

Processes Influencing the Behavior and Fate
of Herbicides in the Environment

Degradation processes are characterized by the
splitting of the herbicide (HB) molecule.
Transfer processes are characterized by the
herbicide (HB) molecules remaining intact
(From Weber, et al., 1973).

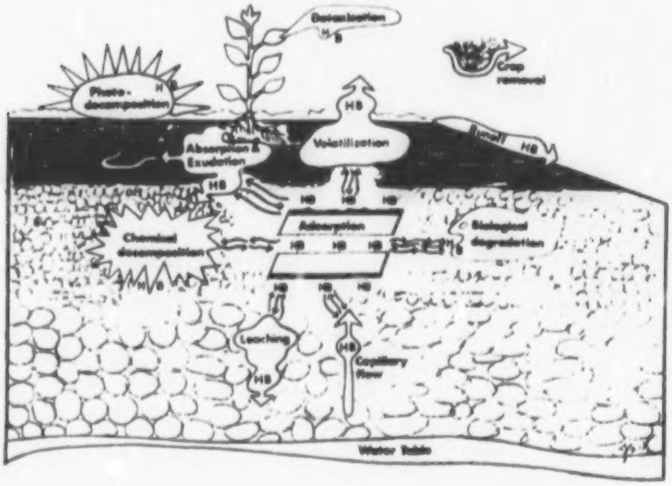


Photo-decomposition
Absorption & Exudation
Volatilization
HB
Crop removal
Runoff
Adsorption
Chemical decomposition
Biological degradation
Leaching
Capillary flow
Water Table

comparing the environmental hazard potentials of various pesticides [Weber, 1975]. Basically, the scheme depends on obtaining information concerning each of the chemical's mobility, longevity, toxicity, and biomagnification effects in the environment, weighing and rating each of these factors, and obtaining their product. The product of these four factors is then used as an index of each chemical's environmental pollution potential.

Mobility of a chemical in the environment depends upon its chemical properties and the properties of the soil. Chemicals with high vapor pressures, such as fumigants and some nematocides, are not included in this scheme. Mobility of a chemical depends upon the ionizability, water solubility, and presence of complexing elements. Chemicals which are cationic in nature or very low in water solubility are classified as immobile. Those which are basic in nature or possess complexing elements such as arsenic or phosphorus are classified as intermediate. Chemicals which are acidic or anionic or have high water solubilities are classified as mobile. The following scheme is used:

Mobility Classification*

1. immobile	cationic; water solubility less than 5 ppm; presence of elements such as Mn, Zn, Hg, Cu, Sn, Ca, K, or NH_4.

FIGURE 2

Average Longevity for 200 Commonly Used Pesticides.

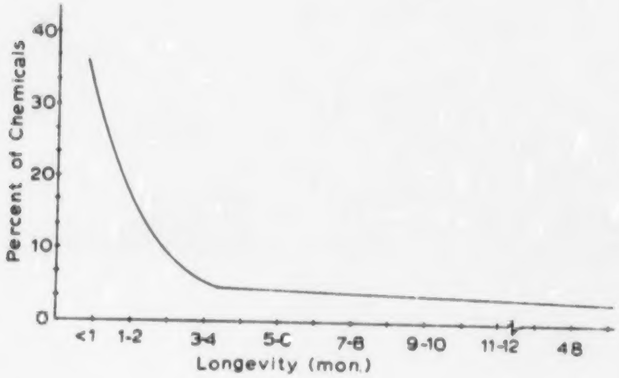
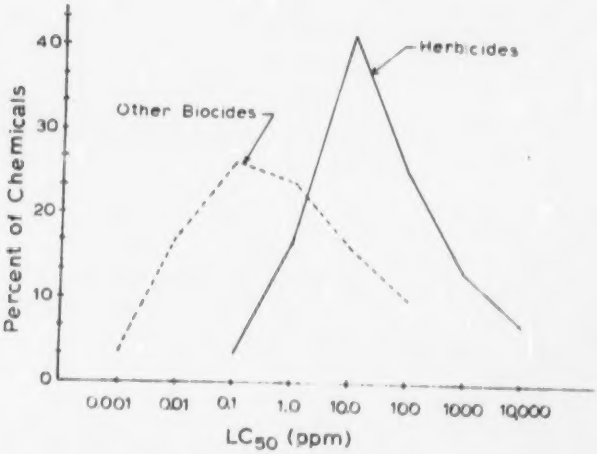


FIGURE 3

Average LC_{50} Values for Herbicides and Other Biocides (Pesticides) to Rainbow Trout or Bluegill.



2. intermediate: basic; water solubilities between 5 and 500 ppm; presence of elements such as P or As.

3. mobile: acidic or anionic; water solubility greater than 500 ppm.

*(A chemical which fits into several categories is placed into the one which best fits all of its properties.)

Longevity of a chemical is measured by how long a chemical controls a given pest, or how long it lasts in the environment. Such information is provided by the manufacturer or by scientists using it in laboratory or field studies. It is generally expressed in terms of length of control or half-life values. It should be pointed out that the majority of the pesticides have half-lives less than four months [Figure 2]. The following classification scheme is suggested.

Longevity Classification

1. readily degradeable: chemical lasts less than 1 month
2. moderately degradeable: chemical lasts from 1 to 6 months
3. slowly degradeable: chemical lasts from 6 to 18 months
4. persistent: chemical lasts longer than 18 months

Toxicity in my scheme refers to the toxicity of the chemicals to fish, primarily rainbow trout and/or bluegill. This information is generally expressed as the LC_{50} value, i.e., the lethal concentration required to reduce the fish population by 50 percent in a given time period (24, 48, or 96 hours). It should be pointed out that the toxicity to fish of herbicides is generally considerably lower than for other pesticides [Figure 3]. The following classification scheme is suggested:

Toxicity Classification

1. nonhazardous: chemicals with LC_{50} values greater than 100 ppm.
2. slightly hazardous: chemicals with LC_{50} values between 1 and 100 ppm.
3. hazardous: chemicals with LC_{50} values between 0.01 and 1 ppm.
4. toxic: chemicals with LC_{50} values less than 0.01 ppm.

Biomagnification is a difficult parameter to measure. It is an indication of whether or not a chemical accumulates in an organism. Organisms such as oysters or fish are placed into specified concentrations of a selected chemical. After a specified period of time, generally 24, 48, or 96 hours, the organism is analyzed for its content of chemical and the ratio of that present in the organism over

TABLE 1

Potential Environmental Hazard Values for Selected Pesticides*

Common Name	Hazard Factors				PEH Values
	Mobility	Longevity	Toxicity	Biomagnification	
Chlorinated hydrocarbons					
aldrin (i)	1	4	4	4	64
BHC (i)	2	4	3	3	72
chlordane (i)	1	4	4	4	64
DDT (i)	1	4	4	4	64
dieldrin (i)	1	4	4	4	64
endrin (i)	1	4	4	3	48
heptachlor (i)	1	4	4	4	64
lindane (i)	2	4	3	3	72
methoxychlor (i)	1	4	4	4	64
mirex (i)	1	3	2	4	24
toxaphene (i)	1	4	4	4	64
"Permanent" pesticides					
ethylmercury chloride (f)	1	4	2	4	32
diquat (h)	1	4	2	1	8
paraquat (h)	1	4	1	1	4
Organaphosphorus pesticides					
acephate (i)	2	1	1	1	2
azinphosethyl (i)	2	1	3	1	6
chlorpyrifos (i)	1	1	3	3	9
coumaphos (i)	2	1	3	1	6
dichlorvos (i)	2	2	3	1	12
dioxathion (i)	2	1	3	1	6
disulfoton (i)	2	1	3	1	6
malathion (i)	2	1	3	1	6
methyldemeton (i)	2	2	2	1	8
methylparathion (i)	2	1	2	1	4
mevinphos (i)	2	2	3	1	12
parathion (i)	2	2	3	3	36
phorate (i)	2	1	4	1	8
TEPP (i)	2	2	3	1	12

TABLE 1 (continued)

Common Name	Hazard Factors				PEH Values
	Mobility	Longevity	Toxicity	Biomagnification	
Carbamates					
carbaryl (i,f)	2	1	2	1	4
carbofuran (i,f)	2	2	3	1	12
s-Triazines					
ametryn (h)	2	2	2	1	8
atrazine (h)	2	3	2	1	12
cyanazine (h)	2	2	2	1	8
prometon (h)	2	4	2	1	16
simazine (h)	2	3	2	1	12
Organic acids					
dalapon (h)	3	1	1	1	3
dicamba (h)	3	2	1	1	6
endothall (h)	3	1	2	1	6
MCPA (h)	3	2	2	1	12
picloram (h)	3	3	2	1	18
silvex (h)	3	2	2	1	12
TCA (h)	3	2	1	1	6
2,4-D (h)	3	1	1	1	3
2,4,5-T (h)	3	2	3	1	18
Miscellaneous					
alachlor (h)	2	2	2	1	8
bromacil (h)	3	3	2	1	18
dichlobenil (h)	2	2	2	2	16
diuron (h)	2	3	2	1	12
monuron (h)	2	2	2	1	8
propachlor (h)	3	2	2	1	12
pyrazon (h)	2	2	2	1	8

f = fungicides; h = herbicide; i = insecticides that present in the water is calculated. This value has been termed the biomagnification or accumulation factor. The following classification system is suggested:

Biomagnification Classification

1. nonaccumulative: chemical concentration in the fish is the same or less than that present in the organism, and the chemical concentration in the organism decreases when the organism is placed into fresh water.

2. slightly accumulative: organism accumulates chemical to less than 10 x that present in the water.

3. moderately accumulative: organism accumulates chemical to between 10 and 1000 x that present in the water.

4. highly accumulative: organism accumulates chemical to greater than 1000 x that present in the water.

The potential environmental hazard of a chemical is dependent on all of the above factors. A useful index is to use the product of the above factors, wherein a completely safe chemical would have a Potential Environmental Hazard (PEH) value of 1, e.g., mobility = 1, longevity = 1, toxicity = 1, biomagnification = 1, and 1 x 1 x 1 x 1 = 1. The most potentially hazardous chemical would, therefore, have a PEH value of 192, i.e., the product of 3 x 4 x 4 x 4 = 192. A list of the PEH values for selected pesticides is shown in Table 1.

PEH values for the chlorinated hydrocarbon insecticides range from 48 to 72, with the majority having values of 64 [Table 1]. Ethylmercury chloride is the only permanent pesticide shown with a PEH value of 32, which is considerably lower than the chlorinated hydrocarbons. The organophosphorus and carbamate pesticides have PEH values ranging from 2 to 36, with the majority having values ranging from 6 to 12. The herbicides listed have PEH values ranging from 3 to 18 and are thus a potential environmental hazard to about the same extent as the organophosphorus and carbamate pesticides.

Table 1 shows that all of the newly developed pesticides are much less potentially hazardous water pollutants than the chlorinated hydrocarbons and the permanent pesticides.

SUGGESTIONS FOR REDUCING HAZARDS FROM AGRICULTURAL CHEMICALS

The biggest step in reducing potential environmental hazards from agricultural chemicals has already been put into effect--that of legally requiring that newly developed pesticides be environmentally safe. Several practices which are recommended to decrease potential hazards from pesticides include: (1) alternative pesticides; (2) optimize pesticide placement; (3) crop rotations; (4) pest-resistant crop varieties; (5) optimize pesticide formulation; (6) mechanical control methods; (7) eliminate excessive treatments, and (8) optimize time of day for pesticide application.

When one or more chemicals of comparable cost and effectiveness are available, use the pesticide with the lowest Potential Environmental Hazard (PEH) value.

Apply herbicides in a narrow rather than wide band. Lightly incorporate them to decrease runoff loss. Apply insecticides in the seed row in row crops rather than broadcast.

Crop rotations can act to suppress insects, weeds, and plant diseases and to decrease the amounts of pesticides required. They also can reduce the amount of eroding soil and washoff.

Use of crops that are resistant to attack by pests will reduce the need for chemical treatments.

Addition of surfactants can improve the efficiency of certain pesticides and decrease the amount required. Some formulations are less mobile in the environment and less toxic to fish and other aquatic life. Granular formulations are generally preferable to liquids because application losses are lower.

Mechanical weed-control operations such as tillage, mulching, or burning reduce the amount of herbicide required, but in some cases may cause increases in soil erosion and washoff. Such practices may also be less economical in some cases.

Control of above ground insects always should be based on plant damage and actual pest counts and should be used only where economic losses justify the pesticide application. Spot spraying of infested areas, rather than spraying of entire fields, should be done. Applications of excessive amounts of pesticides as an insurance against pests should be avoided.

Practices which would help to minimize water pollution from agricultural fertilizers would include: (1) eliminate excessive fertilization, (2) apply nitrogen forms that are the least mobile, (3) time nitrogen application or apply smaller amounts several times, (4) use crop rotations containing legumes, (5) use animal wastes for fertilizer, and (6) plant winter cover crops.

Soil tests should be utilized in making fertilizer recommendations. Only enough fertilizer should be applied to produce the crop.

Ammonium, ammonia, and urea forms of nitrogen are less mobile in soils than nitrate forms and could reduce runoff and leaching losses. Nitrogen fertilizer should be applied at the time of greatest need by the crop. This may be three to four weeks following emergence of the crop--corn, for example. Split applications of nitrogen may be used to decrease losses, e.g. some applied at planting and some as a summer sidedress.

Crop rotations in which legumes are included reduce the amount of nitrogen required. Deep-rooted crops reduce nitrate leaching. Sod crops reduce soil erosion and washoff.

Application of animal wastes as fertilizer improves the soil physical properties, decreases the amount of commercial fertilizer required, and provides a source of nitrogen available all season long.

Winter cover crops can reduce nitrate leaching and soil erosion and runoff.

Another important aspect of reducing water pollution from agricultural chemicals is the use of sound erosion-control practices. Practices which help to keep the soil from eroding include: (1) no-till planting; (2) conservation (minimum) tillage; (3) contouring or terracing the land; (4) contour strip-cropping; (5) grassed waterways and outlets, and (6) sod-based crop rotations.

Practices such as no-till and conservation (minimum) tillage greatly reduce the amount of runoff and soil-erosion losses for most soils. Pesticide and nutrient losses should likewise be reduced, but in some cases such practices create pest problems and higher applications of pesticides are required. More research is needed for optimum utilization of these practices.

Contour farming and strip-cropping greatly reduce soil erosion and washoff of water containing nutrients and pesticides.

Grassed waterways and outlets and sod crops reduce soil erosion and runoff and greatly decrease water pollution.

In summary, the application of agricultural chemicals needs to be optimized. Sound soil-conservation practices must be employed to conserve soil and water and to keep pesticides and nutrients out of the streams and waterways.

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A QUANTITATIVE STUDY OF THE NUTRIENT, SEDIMENT, AND COLIFORM BACTERIAL CONSTITUENTS OF WATER RUNOFF FROM THE RHODE RIVER WATERSHED

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ABSTRACT

Water discharge rates, volume-integrated concentrations of nitrogen, phosphorus, and mineral particulates, and total and fecal coliform bacterial concentrations at discrete times were monitored for a year on five watersheds. These watersheds vary in area from 29 ha to 254 ha and they drain into the Rhode River, a subestuary of Chesapeake Bay. The watersheds have been mapped in detail with respect to land use, and nutrient and sediment data have been analyzed to give mean seasonal area yield loading rates for each land use category. Total and fecal coliform concentrations were found to be highly correlated. High correlations were found between coliform discharge rates and water discharge rates for a given season from a given watershed, but this relationship was different for the other seasons and for the other watersheds.

INTRODUCTION

The Rhode River is a subestuary of Chesapeake Bay, located just south of Annapolis, Md. The Chesapeake Bay Center for Environmental Studies, established in 1965 by the Smithsonian Institution, is a 2,600-acre research facility on the watershed of the Rhode River. The scientific research at the Center is designed to attempt an ecosystem analysis of this small subestuary. Our definition of this estuary includes interactions with its watershed and airshed as well as with Chesapeake Bay proper. The land in the watershed of the Rhode River falls into a mixture of land use categories typical of the western shore of Chesapeake Bay.

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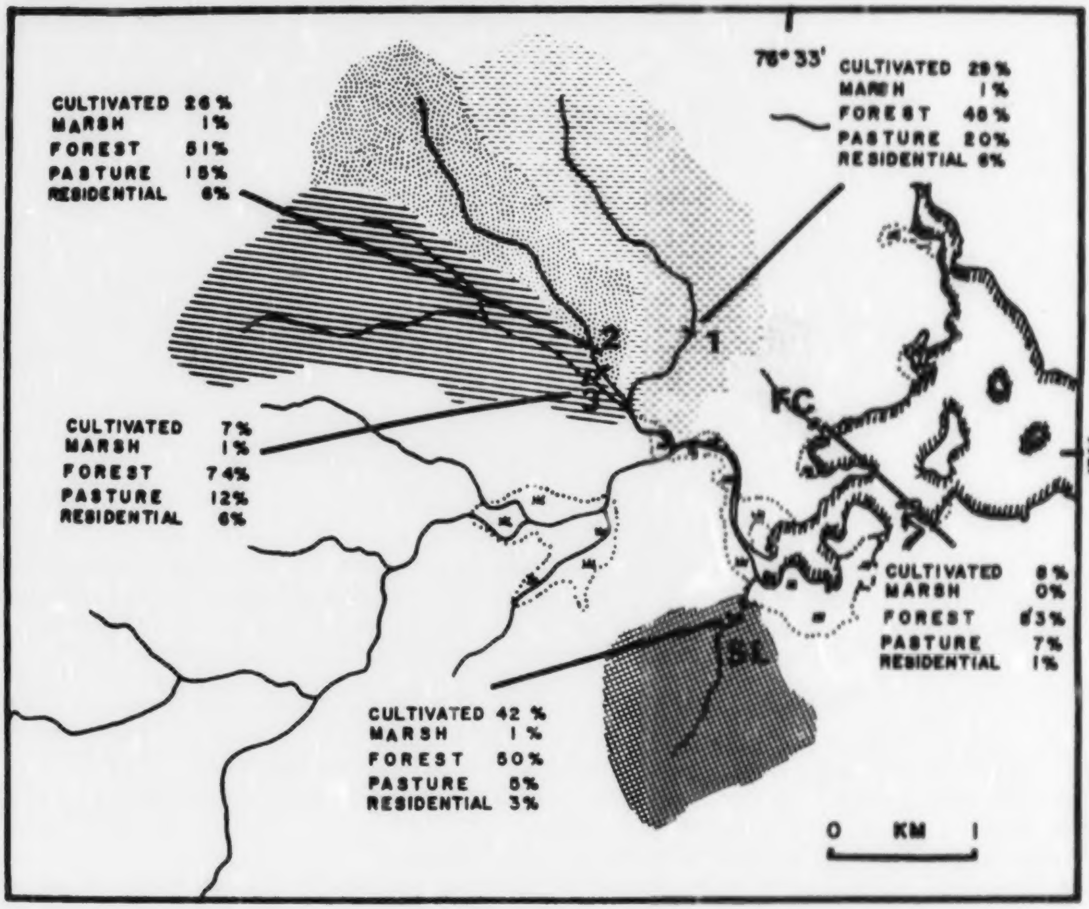
FIGURE 1

Map of the Five Monitored Subwatersheds of the Rhode River

A bar connects a table of the land use for each basin to the location of the weir for that basin. The full names of the basins and their abbreviations are as follows: North branch of Muddy Creek (#1), Blue Jay branch of Muddy Creek (#2), Williamson branch of Muddy Creek (#3), Steinlein branch of Muddy Creek (SL), and Fox Creek (FC).


76° 33'
CULTIVATED 29 %
MARSH 1 %
FOREST 46 %
PASTURE 20%
RESIDENTIAL 6%
CULTIVATED 26 %
MARSH 1%
FOREST 51%
PASTURE 15%
RESIDENTIAL 6%
2
1
3
FC
CULTIVATED 7%
MARSH 1%
FOREST 74%
PASTURE 12%
RESIDENTIAL 6%
38° 53'
CULTIVATED 9 %
MARSH 0%
FOREST 83%
PASTURE 7%
RESIDENTIAL 1%
SL
CULTIVATED 42 %
MARSH 1 %
FOREST 50%
PASTURE 5%
RESIDENTIAL 3%
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A recent review on non-point sources of pollution points out the lack of data in the form of area yield rates rather than concentration units [Loehr, 1974]. Some investigators [e.g. Hobbie and Likens, 1973] have attempted to utilize regressions of concentrations of non-point pollutants versus water discharge rates in order to calculate pollutant discharge rates. The low correlations obtained indicated that this approach is of little value. The reasons for the low correlations and the correct approach to the problem have been reported previously [Nelson, 1970; Henderson, et al., in press]. Volume-integrated sampling, where possible, provides concentrations of parameters representative of the average of all the water discharged during a given time period. No guessing or assumptions of concentrations between sampling times are involved, only the usual problems of experimental error inherent in any research.

The purpose of this report is to summarize the more important results from the first year's measurements of runoff from five watershed basins which drain into the Rhode River. Volume-integrated samples have been taken at weekly time intervals for analysis of nutrients and minerals. All parameters were analyzed on a land use basis and values are expressed in area yield units. We hope that this will result in some answers to such questions as the magnitude of loading due only to leaching from soils of currently undisturbed areas, and the extent to which various agricultural practices contribute to non-point source pollution.

METHODS

Land Use Mapping

Colored aerial photos, taken in June, 1972 by the National Aeronautics and Space Administration, were corrected for optical distortion with a Bausch and Lomb Zoom Transfer Scope. They were then projected at a scale of 1:12,000 upon detailed topographic maps (2 and 5-foot contour lines) prepared by the Aero Service Corporation. Tracings were then made of land use boundaries in each basin on clear acetate. The area of each land parcel was measured five times with a Keuffel and Esser Model 620015 compensating polar planimeter. The mean values were used. The land parcels were classified under 16 categories. For this report, these have been reduced to five categories: (A) cultivated croplands; (B) wet areas (this includes fresh water marshes, swamps, and ponds); (C) forest and brush lands (includes various stages of forest succession); (D) pastureland and other grasslands, and (E) nonsewered residential and other disturbed areas (roads, bare ground, dumps, commercial, institutional and paved areas). In Figure 1 the basins are shown along with the proportion of each land use on each watershed and the location of the weirs. Figure 2 is a detailed map of the land use parcels on one watershed (North Branch, Wier 1).

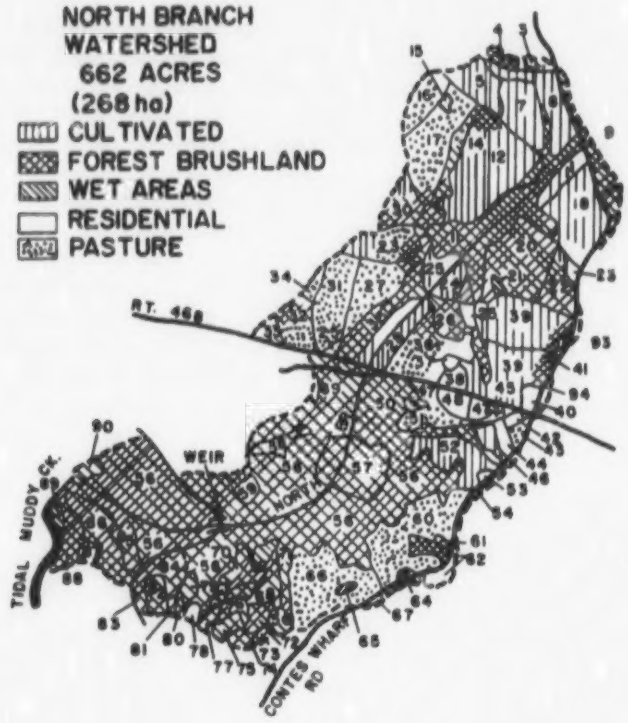
Population and Fertilizer Data

Data on animal populations and chemical fertilization practices were obtained by questioning each farmer on the watersheds in the summer of 1973. Human populations were estimated in 1973 by multiplying the number of year-around telephones by 3.2.

FIGURE 2

Detailed Land Use Map of the North Branch of Muddy Creek Watershed (Weir #1)

Numbered areas are individual land use parcels, discernable from aerial photographs.


NORTH BRANCH
WATERSHED
662 ACRES
(268 ha)
▥ CULTIVATED
▩ FOREST BRUSHLAND
▨ WET AREAS
☐ RESIDENTIAL
▦ PASTURE
RT. 468
WEIR
NORTH
TIDAL MUDDY CK.
CONTES WHARF RD
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Design of Weirs and Sampling Methods

All five weirs are sharp-crested V-notch weirs. The Fox Creek weir has a 45 cm deep, 90° V-notch. All others have 85 cm deep, 120° V-notches. All have instrument buildings with stilling wells. Water discharge was monitored for hydrological purposes with a Leupold and Stevens Model 7001 depth recorder. Digital data was transferred to paper punch tape every 15 minutes. Time and rate of water discharge was also measured with a Leupold and Stevens Model 61R flowmeter equipped with a sampling switch. Each pulse from the sampling switch actuated a sampling cycle by triggering a recycling interval timer which controlled a peristaltic pump. The pump drew water from a point just upstream from the bottom of the V-notch and deposited it into a 201 plastic container. The pumps were protected from freezing by placing them in insulated boxes with thermostated heater strips. These samples are called "volume-integrated." "Grab samples" were also taken at designated times of the year [Table 2] at each weir manually in steril bottles for analysis of bacterial parameters. The time and discharge rate at each weir were recorded, when such samples were taken.

Nutrient Analyses

Volume-integrated samples were analyzed for total phosphorus colorimetrically by reaction with ammonium molybdate and reduction with stannous chloride [American Public Health Association, 1971] after digestion with perchloric acid [King, 1932]. Volume-integrated samples were analyzed for total Kjeldahl nitrogen by Nesslerization after digestion and distillation of ammonia [Martin, 1972]. Volume-integrated samples were analyzed for nitrate plus nitrite-nitrogen by reduction to nitrite on a cadmium amalgam column [Strickland and Parsons, 1965] and subsequent coupling to sulfanilamide [American Public Health Association, 1971]. Total nitrogen was calculated as the sum of the total Kjeldahl nitrogen plus nitrate and nitrite-nitrogen.

Mineral Sediment Analyses

Volume-integrated samples were filtered through Millipore, 0.45 μm pore size, pretreated membrane filters. Mineral particulates were determined gravimetrically [Banse, Falls and Hobson, 1963] after oxidation of organic matter with 30 percent hydrogen peroxide [Pierce, Nelson and Colquhoun, 1972].

Enumeration of Coliform Bacteria

Samples were collected from the surface of the water running through the weirs. The multiple-tube fermentation technique of three parallel test tube dilution series was used in estimating total coliform numbers and the elevated temperature test in estimating fecal coliform densities, respectively [American Public Health Association, 1971]. The assay results were expressed as most probable numbers [(MPN)/100 ml].

Land Use Matrix Calculations

Each piece of concentration data for a given time period (volume-integrated samples) for a given parameter was multiplied by the volume of water discharged during that same time and then was inserted into the appropriate one of five simultaneous equations. For example, the following total phosphorus land use analysis was for days 59-148 of 1974. The letters A through E represent the area yield rates from the different land use categories. The numbers before each letter are the number of hectares of that land use on that watershed.

Equations

Weir 1 $69.7\,A + 1.9\,B + 107.7\,C + 46.9\,D + 13.2\,E = 47\ \text{KgP}$

Weir 2 $47.2\,A + 2.0\,B + 92.6\,C + 27.3\,D + 11.0\,E = 29\ \text{KgP}$

Weir 3 $18.2\,A + 0.45\,B + 188.5\,C + 31.8\,D + 15.5\,E = 71\ \text{KgP}$

Weir SL $62.0\,A + 0.36\,B + 72.9\,C + 7.4\,D + 4.1\,E = 38\ \text{KgP}$

TABLE 1

Mean Seasonal Area Yield Loading Rates for Five Land Use Categories on the Rhode River Watershed in 1974

Land Use Category	Total - N		Total - P		Mineral Particulates	
	gN/ha day	gN/ha cm	gP/ha day	gP/ha cm	Kg/ha day	Kg/ha cm
(Winter)						
Cultivated crops	5.9	28	1.2	6.7	0.34	1.7
Wet areas	-810	-3,800	2.2	11	1.4	7.2
Forest	2.1	9.9	1.3	6.0	0.19	0.94
Pastureland	11	52	1.5	6.9	0.097	0.48
Residential	44	190	-1.1	-6.1	0.91	5.6
(Spring)						
Cultivated crops	31	96	4.7	14	0.54	2.5
Wet areas	-640	-2,000	30	93	-20	-60
Forest	1.5	4.5	0.064	0.20	0.13	0.40
Pastureland	20	62	4.8	18	0.49	2.3
Residential	81	250	57	180	10	31
(Summer)						
Cultivated crops	9.0	3.5	3.3	13	0.59	2.3
Wet areas	-160	-640	-85	-330	-13	-48
Forest	0.51	2.0	0.18	0.68	0.028	0.11
Pastureland	2.4	9.3	2.0	12	-0.058	-0.22
Residential	9.3	36	14	55	1.9	6.9
(Fall)						
Cultivated crops	1.7	7.1	0.094	0.48	0.028	1.2
Wet areas	-19	-64	15	70	2.6	12
Forest	0.97	4.0	1.3	6.0	0.13	0.60
Pastureland	6.3	26	3.9	18	0.33	1.50
Residential	-20	-84	-10	-51	-1.1	-10
(Entire Year)						
Cultivated crops	12	46	2.3	9.2	0.38	1.5
Wet areas	-400	-1,500	-11	-42	-7.4	-29
Forest	1.2	4.9	0.70	2.8	0.16	0.63
Pastureland	9.7	38	3.0	14	0.21	0.82
Residential	28	110	15	58	2.9	11.3

Weir FC $2.35\ A + 0.\ B + 24.39\ C + 2.21\ D + 0.43\ E = 2.8$ KgP

Solutions

A = cultivate = 0.418 KgP/ha = 4.7 gP/ha day

B = wet areas = 2.67 KgP/ ha = 30 gP/ha day

C = forest brush = 0.0057 KgP/ha = 0.064 gP/ha day

D = pasture = 0.43 KgP/ha = 4.8 gP/ha day

E = residential = 5.07 KgP/ha = 57 gP/ha day

RESULTS

Nutrients

Mean seasonal area yield loading rates for total nitrogen and total phosphorus are given in Table 1. The values are expressed in terms of weight per area per day and also as weight per area per rainfall. The first is more useful in a specific evaluation of sources and magnitudes of pollution for Rhode River in 1974. The second might be more useful for modeling purposes. The seasons were arbitrarily defined as: (I) Winter--December, January, and February; (II) Spring--March, April, and May; (III) Summer--June, July, and August; (IV) Fall--September, October, and November. Overall, for the entire year, the Residential category had the highest area yield rate for both phosphorus and nitrogen. This was true also for every season except fall. Cultivated cropland had the second highest rate for phosphorus and its contribution was most in the spring and least in the fall. Pastureland had the second highest rate for nitrogen and its rate peaked in the spring. The rates for forest were always low, and wet areas had negative overall yearly rates for both nitrogen and phosphorus. When water discharge rates were high, wet areas had high positive rates for phosphorus, and when water discharge rates were low, these areas had high negative rates. Overall, cropland contributed 5 times as much nitrogen as phosphorus to runoff but the ratio was only 2.7 in the summer while it was 18 in the fall. Pastureland contributed 3 times more nitrogen than phosphorus to runoff for the year, but the ratio was 7.3 in the winter and only 1.2 in the summer. The residential and forest categories contributed about twice as much nitrogen as phosphorus to runoff.

Mineral Particulates

Mean seasonal area yield loading rates for mineral suspended particulates are given in Table 1. The residential category had the highest loading rate for the year, followed by cultivated cropland, pastureland, and forestlands. The wet area category had a negative rate overall for the year but had high positive rates in the fall and winter. The residential category had its highest rate in the spring and cultivated cropland contributed most in the spring and summer; both had their lowest rates in the fall.

TABLE 2

Estimated Total and Fecal Coliforms in Surface Water Samples Collected at Designated Weirs

Days of 1974	Weir	Water flow (liters x 10^6 /day)	Coliforms		Days of 1974	Weir	Water flow (liters x 10^6 /day)	Coliforms	
			total (MPN/100 ml)	fecal				total (MPN/100 ml)	fecal
52	1	1.13	170	7	294	1	0.100	1500	1500
	2	0.94	130	27		2	0.030	11000	2400
	3	1.28	280	180		3	0.000	--	--
	SL	0.76	94	33		SL	0.000	--	--
	FC	0.12	17	4		FC	0.000	--	--
77	1	2.37	130	130	302	1	0.50	1500	750
	2	1.72	540	170		2	0.006	1500	150
	3	2.59	110	46		3	0.000	--	--
	SL	1.28	17	13		SL	0.000	--	--
	FC	0.25	22	22		FC	0.003	240	90
105	1	3.21	79	11	308	1	0.080	2400	1100
	2	2.44	130	17		2	0.006	460	460
	3	--	79	33		3	0.000	--	--
	SL	0.74	33	11		SL	0.000	--	--
	FC	0.27	49	33		FC	0.006	11000	4600
133	1	3.08	2400	2400	315	1	0.050	200	150
	2	2.16	2400	2400		2	0.020	40	<30
	3	2.95	920	920		3	0.000	--	--
	SL	1.87	350	220		SL	0.009	430	70
	FC	0.15	920	280		FC	0.010	40	<30
168	1	1.06	24000	11000	322	1	0.100	75	39
	2	0.30	1500	1100		2	0.050	23	3
	3	0.33	1100	1100		3	0.000	--	--
	SL	0.43	4600	750		SL	0.040	93	15
	FC	0.04	4600	1100		FC	0.009	93	43
259	1	0.010	93	93	336	1	2.07	460	460
	2	0.000	--	--		2	2.17	460	460
	3	0.000	--	--		3	1.52	2400	2400
	SL	0.000	--	--		SL	1.43	2400	460
	FC	0.001	2400	2400		FC	0.070	240	93
273	1	0.330	4600	1100	350	1	12.3	2400	210
	2	0.090	11000	460		2	18.9	2400	1100
	3	0.000	--	--		3	14.7	28	28
	SL	0.000	--	--		SL	29.2	2400	240
	FC	0.002	2400	2400		FC	0.510	2400	1100

Coliform Bacteria

Total and fecal coliform bacteria were measured as indexes of pollution from animal origins. Concentration estimates and water discharge rates for a series of days in 1974 are given in Table 2. The product of these two sets of values would be the rates of bacterial discharge on those days. Most probable numbers of total coliforms ranged from 3 to 24,000/100 ml when there was water flow. In general, coliform concentrations increased as water discharge rates decreased up until late summer, when the creeks dried up. In the late fall and winter coliform concentrations again increased at the same time as did the water discharge rates, so that a maximum bacterial discharge rate for the year occurred in late December. Thus on day 133 [Table 2], the combined five watersheds (850 ha) discharged 1.0×10^4 m^3 of water with a weighted mean total coliform concentration of 1,000 MPN/100 ml for a discharge of 1.6×10^{11} total coliforms/day. On day 350, the five watersheds discharged 7.6×10^4 m^3 with a weighted mean total coliform concentration of 2,000 MPN/100 ml, for a discharge of 1.5×10^{12} total coliforms.

A high correlation was found between log total coliform concentration and log fecal coliform concentration [Figure 3]. A least squares regression gave (log total coliform MPN/100 ml) = (0.70) (log fecal coliform MPN/100 ml) + (0.87) (with r = 0.82). The values for each weir seem to fall into the same general relationship, rather than segregating into separate clusters. The calculation of overall loading rates between times of assay is difficult. A regression of log fecal coliform concentration versus log water discharge rates, using all time points and data from all weirs had a correlation coefficient of 0.48. If data from individual weirs were analyzed separately, only weirs 2 and SL had better correlation coefficients (r = 0.72 and 0.77, respectively). When only data from November and December were compared, a higher correlation was found. When data from all weirs for that time period were analyzed, the result was (log fecal coliforms discharged/ha day) = (0.25) (log liters of water discharged x 10^6/ha day + (1.11) (with r = 0.80). If data from individual weirs for this time period were analyzed separately, the correlation coefficients varied from r = 0.60 to 0.95. Weirs 2 and FC gave identical regressions of results; (log fecal coliforms discharged/ha day) = (0.31) (log liters discharged x 10^6/ha day) + (1.26) (with r = 0.92 and 0.60, respectively). However, SL weir gave a relationship of (log fecal coliforms/ha day) = (-0.096) (log liters water discharged x 10^6/ha day) + (1.29); (with r = 0.95). Thus it would seem that there are basins with coliform discharge properties which fall into different classifications and, at least for a given season at a given weir, a high correlation existed between water discharge and coliform discharge. The similarities between watersheds 2 and FC with regard to coliform discharge, were not land use [Figure 1], or domestic animal population densities. Coliform concentrations in runoff waters had very low correlations with concentrations of nitrate, organic nitrogen, total phosphorus, or sediment concentrations.

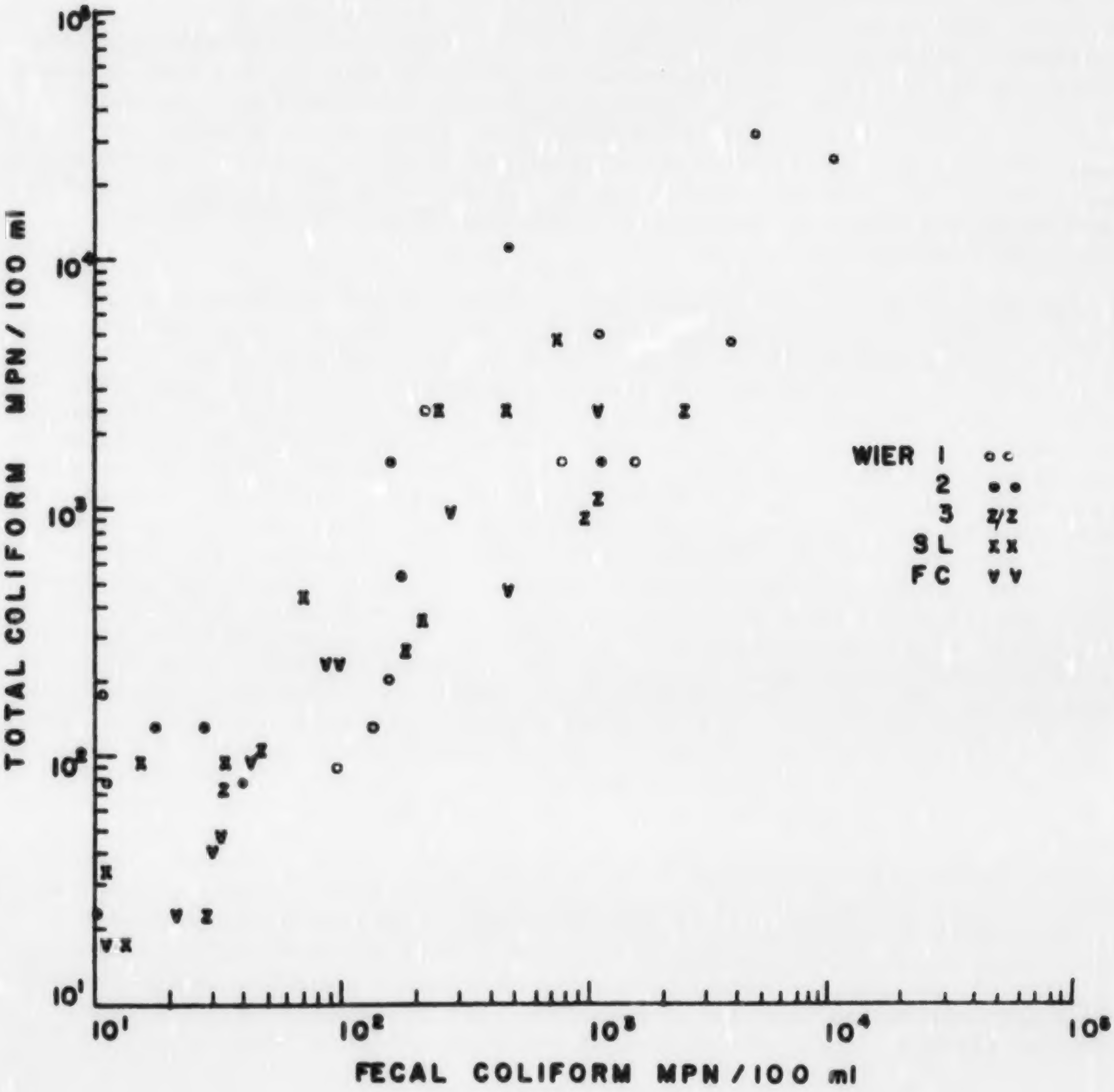
DISCUSSION

The five watersheds studied in this report have slopes which average about five percent. They have sedimentary, easily erodible soils which vary primarily with respect to their geological age rather than to their mineral composition. The results in this report are preliminary, since they were obtained during

FIGURE 3

Correlation Analysis of Total and Fecal Coliform Concentrations

(Units are most probable numbers of coliforms per 100 ml).



a one-year study and have not been subjected to extensive analysis. As time permits and more data are obtained over the course of several years, the conclusions reached in this report may become more precise and more generally applicable to other times and places. The results in Table 1 are most accurate for the entire year. As the time period is shortened, and less data points are available, the rate values begin to show more noise. For this report we chose to report only seasonal and yearly rates. Some interesting facts can be derived from the data in Table 1. Cultivated cropland had an average yearly area yield loading rate to the Rhode River of 12 g nitrogen/ha day, while pasturelands average 9.7. Rainfall on the Rhode River watershed delivered an average of 20 g nitrogen/ha day [Correll, 1975]. The average farmer on the watershed also applied 178 g nitrogen/ha day to cultivated crops and 145 g nitrogen (as chemical fertilizer)/ha day to pasturelands. Only the residential category had higher yield loading rates of nitrogen than was delivered by rainfall. This indicates that very high rates of cropping and denitrification must be going on most of the time.

In the case of phosphorus, cultivated croplands and pastureland delivered to the estuary yearly average area yields of 2.3 and 3.0 g/ha day, respectively. Rhode River rainfall averaged 1.2 g P/ha day, while the average farmer applied 91 and 52 g P/ha day, respectively. Thus, much more phosphorus is being applied to these lands than is coming off as runoff. It should be noted, however, that the fertilizer applications are made in the spring, rather than evenly throughout the year. Area yield loading rates from cultivated croplands were highest in spring and summer and those from pasturelands were highest in the spring.

When the data on mineral particulate area yield loading rates are examined, two things become apparent: (1) As the watershed becomes urbanized, the loading will increase proportionately, and (2) if wet areas are filled and channeled, less sediments will be trapped before they reach the estuary. Surprisingly, with current agricultural practices on the watershed, cultivated cropland had an average area yield loading rate of only two to three times that of forest, and pasturelands had area yield loading rates not much higher than those found for forest.

Contamination of the Rhode River Estuary with bacteria of fecal origin by land runoff is a serious problem, since a total coliform concentration of 70 MPN/100 ml over shellfish beds is sufficient grounds to prohibit shellfish harvesting. At times of high water discharge, such as days 133, 168, and 350 [Table 2], most of Rhode River was contaminated with concentrations of over 100 MPN/100 ml of coliform bacteria. Since total coliform concentrations correlated highly with fecal coliform levels, it would seem that these two parameters are a result of the same factors. At this time, land use patterns and animal or human population densities do not seem to be the key factors. The use of regressions is of value for the calculation of total discharges of bacteria between times of assay for a given watershed in a given season. In view of the currently not understood differences between watersheds, the regressions cannot safely be extended to other watersheds. The average human populations on the watersheds studied for this report in 1973 were 0.32 persons/ha. Domestic livestock populations in 1973 were 0.42 cattle, 0.034 horses, 0.15 hogs, and 0.15 chickens per ha. These numbers should be multiplied by 7.4 to give populations/ha of pasturelands. Could the watershed runoff be the source of sufficient coliform bacteria to account for observed levels in the Rhode River estuary? If we assume that the discharge rate for the entire 3,260 ha watershed is about equal to the average rate for the 850 ha monitored, then on day 350 [Table 2] 5.8 x 10^{12} coliforms were discharged into the Rhode River. The estuary has a mean low water volume of 9.8 z 10^9 l. It would therefore take only about 1.2 days at this rate to discharge enough coliforms into the Rhode River to give an average concentration of 70 MPN/100 ml. Evans and Owens [1972] estimated that the average survival time for coliforms in land-drainage waters was about 57 days. Thus, land runoff could account for the high concentrations of coliforms observed in the Rhode River even at times of maximum contamination.

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HARVESTING SOUTHERN FORESTS:

A THREAT TO WATER QUALITY?

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INTRODUCTION

Although pending regulations make pollution from forestry activities increasingly significant, the effects of harvesting on water quality are not known. In this paper I shall treat sediment as the primary pollutant and emphasize the southern pinery, which presently occupies more than half the area of southern forest lands and may cover two-thirds in less than 25 years. The South furnished almost half of the nation's wood in 1970 [U.S. Forest Service, 1973]. Since wood demand is expected to double by the end of the century, supplying this need will require a 70 percent increase in annual softwood growth [Southern Forest Resource Analysis Committee, 1969]. Management will intensify, and operations will become increasingly mechanized. Much private forest will be grown in short rotations, clearcut, and reestablished with superior trees. Intensive culture, however, may impair southern soil and water resources. Thus a potential conflict exists between the need to increase pine production at competitive costs and the growing concern for environmental safeguards.

By presenting the following information, I hope to place possible pollution from harvesting into perspective and to focus on questions needing resolution before limited research resources are committed.

HOW SOUTHERN FORESTS INFLUENCE WATER QUALITY

Compared to other forest regions, much of the South is unique because of previous cultivation and widespread erosion of forest lands. Soil and water hazards are most acute on the hilly topography of the Piedmont and the Coastal Plain; after decades of exploitative, row-crop agriculture in these areas, millions of acres were abandoned. Of the 120 million acres of forest land in the Coastal Plain, 22 million have been subjected to accelerated erosion [McClurkin, 1967]. After fire control was achieved, large areas of these lands developed a vegetal cover and gradually reverted to forests, which helped to stabilize the soils. In addition to seriously eroded land that formerly was farmed, the South contains a large acreage of abused and poorly stocked pinelands with fragile soils that possess a high pollution potential. Runoff and sediment from eroding lands contributed to serious downstream damage, covered minor flood plains, and created

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sand-filled channels which are important sources of sediment today.

Recognition of the need to accelerate erosion control and to restore land productivity stimulated massive tree planting. We have some information about how fast favorable hydrologic characteristics were regained, but there are no data showing how future harvesting will affect water quality.

In a review draft outlining water-quality criteria, the Environmental Protection Agency [1973] has proposed 80 mg per liter (ppm)--equivalent to 18 pounds per acre-inch of runoff--of suspended solids in water for protecting fish, wildlife, and recreation resources. However, such standards cannot be meaningful until we know how much sediment is produced by undisturbed forest lands. Some baseline data are available: sediment yields from small Coastal Plain catchments of pole-size loblolly pine planted for erosion control and from mature southern pine seldom exceeded 0.1 ton per acre annually when rainfall ranged from 42 to 66 inches and stormflows were up to 12 area-inches [Ursic and Dendy, 1965]. For small, pine-covered headwater catchments in north Mississippi, annual sediment production averaged 13 pounds per acre-inch of stormflow, and most of this comparatively minor yield was not from forested slopes but from channels created by previous farming [Ursic and Duffy, 1972]. Sediment concentrations were independent of individual stormflow volumes or annual water yields. Therefore, since 13 pounds per acre-inch may approximate the norm for well-stocked, undisturbed southern pine forests of the Coastal Plain, this figure will be utilized here for comparisons with other watersheds and for discussion of the proposed EPA standard.

The small catchments may be improving chemical water quality by accumulating more NO_3-N, NH_4-N, PO_4-P, Ca, and K from rainfall and fallout than are being lost as dissolved constituents in stormflows [Schreiber et al., 1974]. However, the additional nutrients carried from the watersheds by particulate matter and those transported to the water table by percolating waters have not been quantified.

Sediment concentrations from small catchments of southern pine in the Ouachita Mountains of central Arkansas were lower than those in Mississippi; yields averaged only 2.3 pounds per acre-inch of flow during an 8-year period [Rogerson, 1971]. Presumably the difference was due to the greater resistance of mountain channels to erosion.

Channels carrying stormflows from the small catchments normally are covered with pine litter, which prevents erosion. However, unusually large stormflows can flush out the litter and accumulated sediment and increase concentrations to 10 times that of the 80 ppm EPA standard. The rates may remain high while litter reaccumulates.

By looking at progressively larger watersheds in the same soil-physiographical province as the Mississippi pine catchments, we can identify some of the factors that affect the quality of water from forest lands. The water from the small catchments flows to larger channels, which are a major source of sediment in the Coastal Plain since they are typically sand and sensitive to the erosive power of flowing water. The channel's contribution to sediment was demonstrated by testing predictive relationships applicable to 3 to 7-acre catchments on an 88-acre catchment. The predictions were reasonably accurate for water yields, but annual sediment production was 13 times greater than expected [Ursic and Duffy, 1972; Tennessee Valley Authority, 1962]. The additional 175 pounds per acre-inch of runoff was sediment that had accumulated in the channels before the pines were planted. Since the runoff from the reforested slopes was comparatively clean, its additional carrying power picked up sediment from the channel bed.

The average annual sediment discharge from a 1,580-acre watershed of mixed land use (45 percent forested) was 0.58 ton per acre-inch of stormflow; little base flow was involved. This is a rate 89 times greater than the 13 pounds per acre-inch from the small headwater catchments. Total sediment discharge averaged 3 tons per acre per year, 19 percent of which was from streambank erosion. A 3.5 mile reach near the gauge contributed 260 tons of sediment per mile annually, equivalent to 280 ppm--more than three times the proposed E-A standard [Piest and Bowie, 1974].

Further downstream, the comparable figures for a 117-square-mile drainage area were 0.70 ton per acre-inch of runoff, an annual yield of 4.8 tons per acre, and a loading factor of 100 ppm per mile from a 4-mile reach near the gauge. The banks of this reach of dredged channel were eroding at the annual rate of 5,700 tons per mile of channel, more than three times the national average for 300,000 miles of streams with bank-erosion problems [Barnes, 1968]. Thus, if we assume that one-third of this watershed is pine and that harvesting would increase base-rate sediment production 10 times to 130 pounds per acre-inch, the total sediment passing the gauging point would be increased by less than 2 percent; 1.4 miles of dredged channel at the mouth of the drainage would be yielding as much sediment as the entire forested portion of the watershed.

How well these large watersheds represent southern conditions is unknown. Because sediment production is influenced by watershed size, channel conditions, and land use, the hazards of extrapolating from one watershed to another are great.

Piest and Bowie [1974] attributed some of the increases in channel conveyance and streambank erosion to land-use changes that reduced sediment content in the runoff. Thus, stabilizing eroding headwater catchments with pine may increase channel erosion downstream--or, conversely, an increase in sediment production from forestry activities may be nullified downstream.

Partially offsetting the effects of cleaner runoff on downstream channel degradation and streambank erosion of sensitive channels will be a reduction in water yield caused by conversion of hardwoods to pine in much of the South, where about 70 million acres of pine sites support unwanted hardwoods [Anderson, 1974]. Converting 20 million acres to pine and planting an additional 10 million acres of poorly stocked lands are listed as priorities in The South's Third Forest--How It Can Meet Future Demands [1969]. Over much of the South, conversion to loblolly pine will increase the annual interception loss by at least 4 inches, an amount equivalent to between 20 and 25 percent of annual streamflow [Swank et al., [1972]. In situations where streamflow reductions are undesirable--as when water is needed for consumption or pollution control--forest management practices can alleviate the losses [Ursic, 1975]. However, if dilution is rejected as a means of pollution control, reduction in water yield need not concern foresters.

HYDROLOGIC IMPACTS OF HARVESTING AND RELATED PRACTICES

Harvesting increases the potential for sediment production by disturbing the soils and the protective forest floor, but most of the existing data about harvesting pertain to soil movement (the relocation of soils on slopes) and not to sediment production, which occurs only when particulate matter reaches a channel. The formulas available for predicting soil movement on slopes were developed for farmland and may be useful for evaluating forest soil disturbances similar to those that expose or compact agricultural soils. However, the formulas are inapplicable to the undisturbed portions of the forests, which seldom have the overland flow (rainfall in excess of infiltration) on which the predictions are based [Lull and Reinhart, 1972].

Simply removing the pine overstory--without reference to logging disturbances--causes the forest floor to deteriorate rapidly. In one study, soil movement on slopes was evident within two years [Dickerson, 1972]. This phenomenon also was observed on small catchments of upland hardwoods deadened with a herbicide and underplanted with pine. Sediment production was increased by about 400 pounds per acre during the first year [Ursic, 1970]. However, a 75 percent reduction in sediment yield occurred during the fourth through seventh years after planting [Ursic, unpublished data]. Thus long-term reductions in sediment after conversion to pine more than compensate for temporary increases.

Logging practices further increase the erosion hazards. Thinning does little damage and its effects are quickly healed, but clearcutting may affect up to 20 percent of a logged area [Dickerson, 1968].

Skidding creates the most serious disturbances. In one study, soil movement from trails created by uphill skidding on a variety of soils, slopes, and slope lengths in an upland hardwood forest averaged 0.84 ton per acre of trail during the first year and then rapidly declined [Dickerson, 1975]. However, if it is assumed that skidding disturbed a maximum of 20 percent of the area, the amount of soil movement was still only 340 pounds per acre. Stormflow increases persisted--probably because soils in wheel ruts were compacted an average of 20 percent to 1.55 g per cm. In addition to increasing runoff, this density would seriously impair survival of planted or natural seedlings and other vegetation [Duffy and McClurkin, 1974]. The estimated time for recovery of soil physical properties to prelogging conditions was about 12 years.

The only data available on sediment production from harvesting concerns small catchments in the Ouachita Mountains of central Arkansas, where yields after harvesting were 11 times greater than the seven year pretreatment average of 14 pounds per acre per year. Sediment returned to prelogging levels in three years [Rogerson, 1971; Rogerson, 1971a].

There are no comparable data for the Coastal Plain or Piedmont. However, the University of Georgia has an active study in the Piedmont, and scientists at the Forest Hydrology Laboratory at Oxford, Mississippi, recently began measuring the effects of clearcutting and stripcutting on 16 small Coastal Plain catchments in west Tennessee. This study should help determine whether loblolly plantations can be clearcut without renewing accelerated erosion. It is hoped that studying small catchments will provide some of the hydrologic and water-quality data necessary for developing a predictive capability applicable to a wide range of southern forest conditions.

Although logging roads are considered one of the most important sources of sediment in mountainous terrain, they do not constitute a great hazard in the Piedmont and Coastal Plain pine forests, where an adequate network of roads from previous farming already exists, and the roads usually follow favorable terrain. In new construction, large cuts and fills seldom are required, and sediment production from logging roads can be managed with available guidelines for construction and maintenance [Hewlett and Douglass, 1968; Kochenderfer, 1975].

HOW WILL WATER QUALITY STANDARDS BE DETERMINED AND APPLIED?

Public Law 92-500 has set interim goals for control of water pollution: "best practical technology" should be achieved by 1977, and "best available technology" by 1983. Agee [1975] points out that water-quality degradation caused by non-point sources may well emerge as the major barrier to achievement of PL 92-500's 1983 goals as control of point sources of pollution is attained. The law includes a "where attainable" clause that will pertain to standards still to be defined.

This clause is extremely important to southern forest-land managers. For example, if 13 pounds per acre-inch is considered the normal base rate for undisturbed pine lands, achievement of the 18 pound per acre-inch EPA standard for water quality may largely prohibit the harvesting techniques presently used on the hilly pine lands of the South. The problem of evaluating, regulating, and monitoring water quality from southern forest lands is compounded by a fragmented land-use pattern, by the fact that almost three-fourths of the southern forest is divided among more than a million small private holdings, and by the difficulty of determining how much sediment is produced by channel and bank erosion rather than forest lands.

Forest managers cannot plan to meet specific water-quality objectives nor can research be effective until scientists know both the size of the drainage areas and the streamflow characteristics to which proposed standards will apply. Still to be decided are whether the standards will be applied universally regardless of the ultimate use of the water involved, and whether they will be relaxed under the "where attainable" stipulation when goals cannot realistically be achieved. In applying proposed standards, regulators should probably consider average annual sediment production from forest lands, since gross disturbances occur infrequently--at minimum intervals of 20 years--and the effects are short-lived.

I do not wish to imply that excessive sediment production from forest lands should be ignored or condoned. I have attempted to place possible pollution from harvesting into perspective, and to point out questions that need to be answered before limited research resources are committed. The American Forestry Association advocates research designed to find valid solutions to forestry-related pollution problems and urges members to become involved at all levels [Pardo, 1974]. I hope this group will support the needed research so that foresters will help meet the objectives of PL 92-500.

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EROSION AND SEDIMENTATION AS RELATED TO INTENSIVE TIMBER MANAGEMENT IN THE SOUTHEAST

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SUMMARY

The objective of intensive timber management is to maximize fiber production while maintaining site quality. The primary activities that may lead to accelerated erosion and sedimentation are access-system design and location, harvesting, site preparation, and planting. Other causal factors in some instances include fire protection and suppression activities. Whenever accelerated erosion and sedimentation do occur, the forest manager must be concerned with two consequences: degradation of site productivity potential, and stream water quality. In most cases the necessary technology, when coupled with common sense, is available to minimize accelerated erosion and sedimentation following intensive forest activities.

The question of how much sediment delivered to streams is permissible remains unanswered. However, forest-resource managers do have the ability to maintain sediment-delivery rates below those found with good agricultural practices in the same region, even though the forest activities are most frequently on steeper land. In addition, the frequency with which any one site will be disturbed is on the order of once every 15 to 25 years, whereas most agricultural land is disturbed at least once a year.

Although there is no soil loss prediction equation that may be applied to disturbed forest land, familiarity with the Universal Soil Loss Equation [Wischmeier and Smith, 1965] as applied to agricultural fields provides insight into the factors most important in the detachment, transportation, and deposition of soil particles. The factors are vegetative cover, soil erodibility, rainfall energy, land slope, and length and type of conservation practice (such as contour plowing, terracing, etc.). In the Southeast, anywhere from 50 to 75 percent of the total annual rainfall energy occurs during the summer months, when the frequency of high-intensity convectional storms is greatest. If intensive forest activities are planned so only a minimum of mineral soil is exposed during these months, soil particle detachment would be drastically reduced.

Additional possibilities are obvious for reducing sediment production through an understanding of the soil loss equation. Among them are restricting intensive activities where soil erodibility and/or slope are high, and operating along the contour.

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Although the Universal Soil Loss Equation is not applicable to watersheds, application of the concepts of source-area hydrology [Hewlett and Nutter, 1970] will lead to further reduction in the delivery of sediment to the stream channel. Source-area hydrology recognizes the expanding and shrinking nature of the stream channel system during a storm. These expanding channels must "reach out" to collect sediment and deliver it to the perennial channel. The source area, and hence the extensiveness of channel expansion, is a function of basin morphology, antecedent moisture conditions, and the intensity and duration of the storm. The majority of the sediment load in the perennial channel due to accelerated erosion is delivered by the channelized flow in the expanded channel system. Therefore, if the variable source area is identified and major disturbance activities are not permitted in the area, the chance of sediment delivery to the perennial stream channel is greatly reduced.

Among current intensive timber-management practices, access systems rank highest in sediment-production potential, followed in order by site preparation, harvesting, and planting. Access systems often are inadequately designed and improperly located, resulting in high rates of sediment delivery to the stream channel. To further compound the problem, the access system often must be overdesigned to accommodate highway-type vehicles, and frequently these vehicles are operated when the roads are too wet. All of these factors lead to more extensive disturbances and concentrated drainage problems than necessary. Frequently 75 to 90 percent of the sediment delivered to the perennial stream channel is from the access system, and delivery often continues long after the operation is closed down.

Types of site preparation vary with the terrain and agency practice, but all have the common objectives of preparing a planting bed, reducing woody and herbaceous competition, and reducing fire hazard. Sediment delivery to the perennial stream channel can be minimized if a technique is selected that: (1) is compatable with the site; (2) leaves as much organic matter intact over mineral soil as possible: (3) is scheduled to reduce exposure during periods of high rainfall energy, and (4) is restricted in the sensitive source areas.

Harvesting activities and related erosion have been studied intensively over the years. These investigations indicate that the three foremeost erosion-control practices are: (1) keeping skid trails diffused and running upslope; (2) keeping equipment out of stream channels and crossing streams only with improved structures which are removed at the end of harvesting, and (3) maintaining organic layer over mineral soil whenever possible.

Planting activities, when conducted sensibly using conservation practices similar to those used in agriculture, will not cause any increase in erosion.

The question that still remains is, "How much sediment delivery to stream channels can be tolerated?" Sufficient information is available to the forest-resource manager to allow control of erosion and sediment delivery to the perennial stream channel. In many cases the application of common sense is all that is necessary to keep erosion within tolerable limits. Will erosion and sediment controls lead to higher costs? Likely not, particularly if the overall goal of maintaining site quality is fundamental to the management scheme. In fact, many of the practices that go hand in hand with reduced sediment production are less costly in terms of wear and tear on machinery even though there may be a slight increase in operation time.

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FOREST-MANAGEMENT PRACTICES AS RELATED TO NUTRIENT LEACHING AND WATER QUALITY

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INTRODUCTION

The need for a satisfactory environment, so pressing in the cities, also is being recognized in the forests. The public has become increasingly aware of forest management and its effects on the forest and water resources of the East. Many streams originating on forest lands are sources for municipal water supplies and are prime sites for forest recreation. Continuous upgrading of water-quality standards by governmental agencies is a sure sign that methods used to manage timber in the future must assure adequate protection of our watersheds.

Undisturbed forested watersheds are generally recognized as primary sources of high-quality water. The physical and chemical natures of streamflow from these watersheds fluctuate constantly in response to natural stress, but are influenced most by man's activities. The following forest-management practices can have major influences on water quality:

1. Changes in nutrient concentration.
2. Changes in water temperature.
3. Changes in stream water turbidity and sedimentation.

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EFFECTS OF VEGETATION MANAGEMENT ON NUTRIENT CONCENTRATIONS

Mineral and organic nutrients come from three sources: geologic weathering of parent material; atmospheric inputs, and biological inputs. In undisturbed forests, the main nutrient losses are through volatilization to the atmosphere, soil erosion, and leaching of dissolved chemicals. Managed forests are subject to additional losses resulting from timber harvesting and residue removal or treatment [Moore and Norris, 1974]. The quantity of nutrients in streams evidently increases when the rate of decomposition of residues exceeds the uptake by vegetation and the exchange capacity of the soil [Rothacher and Lopushinsky, 1974].

New Hampshire

The effects of clearcutting on nutrient discharge in streamflow attracted widespread attention after experimental results on the Hubbard Brook Experimental Forest in the White Mountains of New Hampshire were published [Bormann et al., 1968; Likens et al., 1970]. Here a 39-acre watershed was completely clearcut; however, all tree and other woody vegetation was left in place. For the next three summers, the entire watershed was sprayed with herbicides to prevent the regrowth of vegetation. It should be emphasized that this experiment was not designed to simulate a normal forest cutting and logging operation, but was used solely for research purposes.

After treatment, concentrations of some nutrients in streamwater were surprisingly high. The concentration of nitrate-nitrogen increased from a maximum 0.5 ppm before treatment to a maximum of 20 ppm after cutting and herbiciding. Concentrations of ions such as calcium, magnesium, and potassium varied from 5 to 30 times above normal [Pierce et al., 1972] while concentrations of sulfate decreased by 3 to 4 ppm [Horbeck et al., 1975].

These increases were attributed to: (1) exposure of the site to greater-than-normal amounts of heat and water, thereby accelerating organic-matter decomposition and subsequent leaching of nutrients; (2) acceleration of the nitrification process, and (3) blocking the uptake of available nutrients that normally would have occurred had vegetation been allowed to regrow [Pierce et al., 1972]. A fourth factor, advanced by Reinhart [1973], is the nature of the podzol soils on the watershed. These soils have large accumulations of organic matter on the surface and mineral horizons that are generally low in available nutrients, and possibly low in their ability to retain nutrients.

Nutrient losses from actual timber-harvest clearcuttings on the White Mountain National Forest in New Hampshire have been found to be much lower. Average nitrate-nitrogen concentrations in streamwater from seven timber-harvest clearcuttings ranged from 1.3 to 4.5 ppm; the maximum recorded was 6.4 ppm. Measurements made downstream below the clearcut areas illustrated the dilution provided by essentially nutrient-free water from areas not cut. Nitrate-nitrogen concentrations in streamwater draining one 160-acre clearcut were 1.5 times those in the stream draining the total drainage area of 240 acres (160 cut, 80 uncut) [Pierce et al., 1972].

A more recent study at Hubbard Brook indicates that nutrient losses can be moderated considerably by strip-cutting. The strip-cutting involved a form of even-aged management for regenerating high-quality yellow and paper birch. An experimental watershed was divided into 80-foot-wide strips, and timber from every third strip was harvested in October, 1970. Plans called for the remaining two series of strips to be harvested in a similar fashion at two-year intervals. Where a strip-cut intersected with a major stream channel, a buffer zone of trees was retained [Hornbeck et al., 1975].

Increases in nitrate-nitrogen occurred immediately after the first strip-cutting and persisted for the 23 months of post-treatment data [Hornbeck et al., 1975]. Increases in mean monthly concentration for the first dormant and growing seasons after logging averaged 0.23 ppm, then increased to an average of 1.30 ppm in the second dormant season. Increases for the second growing season (June-September) averaged 0.52 ppm. Increases of as much as 1.64 ppm were measured in the dormant season when stream nitrate-nitrogen concentrations are usually highest. On forested watersheds at Hubbard Brook, minimum concentrations (less than 0.11 ppm) usually occur in summer and early fall; concentrations increase in late fall as vegetation becomes dormant, and they reach a maximum (0.46 to 1.14 ppm) in early spring.

After cutting, calcium concentrations--which averaged 2 ppm before cutting--increased about 0.8 ppm during the dormant season and about 0.3 ppm during the growing season. Potassium concentrations--which averaged about 0.2 ppm before cutting--increased about 0.1 ppm. Strip-cutting did not affect magnesium concentrations until the fall of the second year after cutting, when concentrations increased about 0.2 ppm over the precutting concentrations of about 0.4 ppm. Concentrations of sulfate in streamwater decreased slightly. These changes are substantially less than results from the clearcut-herbicided watershed at Hubbard Brook [Hornbeck et al., 1975].

West Virginia

In other parts of the East, where soil conditions differ from those in New Hampshire, results contrast strongly with the Hubbard Brook data. In West Virginia, nutrient leaching following a timber-harvest clearcutting was far less than in New Hampshire. The stream in this clearcut watershed was protected by a buffer strip, 33 to 66 feet wide, left along each side of the perennial stream channel. Only the large trees were removed from this area. Dissolved solids in the stream were virtually unaffected by clearcutting. However, minor changes in ionic concentrations did occur. These changes were generally slight, rather irregular, and short-lived [Aubertin and Patric, 1974].

In the first year after harvest, average nitrate-nitrogen concentration in streamwater was only 0.18 ppm for the growing season and 0.49 ppm for the dormant season [Aubertin and Patric, 1974]. Phosphate concentrations showed slight increases while the sulfate concentrations decreased. Clearcutting had negligible effects on calcium, magnesium, sodium, potassium, iron, copper, zinc, manganese, and ammonium-nitrogen. Success in avoiding damage to water quality was attributed to careful road management, retention of a forest strip along the stream, and rapid vegetation regrowth after clearcutting [Aubertin and Patric, 1974].

TABLE 1

Average Nutrient Concentration of Stream Water
on Partially Clearcut and Control Leading Ridge Watersheds
October 1973 - May 1974.

ELEMENT	CONTROL LR-1	TREATED LR-2	CONTROL LR-3
	------------	PPM	----------------
CALCIUM	4.36	2.28	4.32
MAGNESIUM	1.96	1.24	1.53
POTASSIUM	1.18	1.15	1.09
SODIUM	0.97	0.90	1.00
SULFATE	9.16	6.43	7.10
NITRATE - N	0.02	0.10	0.01

North Carolina

At the Coweeta Hydrologic Laboratory in North Carolina, drastic alterations of the forest ecosystem have not caused long-term losses of cations by erosion or leaching by drainage waters. Of the cations studied (Ca, Mg, K, Na), only calcium concentrations were appreciably higher than that of undisturbed forests, and then only on watersheds that received lime applications during their treatment. The highest mean annual concentration of calcium was 1.09 ppm. Maximum concentration of nitrate-nitrogen was 1.23 ppm on a grass-to-forest succession watershed. The second highest concentration observed was 0.25 ppm, after a hardwood-to-white pine conversion, as compared with 0.002 to 0.013 ppm for undisturbed watersheds. Douglass and Swank [1975] concluded that the concentration of all cations and anions on treated watersheds is quite low, and that the increased export of nutrients is insufficient to create pollution problems for drinking water.

Pennsylvania

A three-phase clearcutting experiment on the Leading Ridge Watersheds at the Pennsylvania State University currently is in progress. Vegetative cover consists primarily of uneven-aged oak, hickory, and maple. Phase one involved the removal, during the winter of 1966-67, of all timber on 21.3 acres of the lower portion of 106-acre Leading Ridge Watershed 2. Timber growing on steep slopes with shallow shale soils was left uncut to prevent serious erosion. This clearcut was essentially a complete riparian cut, since all of the main stream channels on the watershed are located within its boundaries. Phase two expanded the clearcut area by 27.0 acres toward the middle-slope portion of the watershed during the winter of 1971-72. During the summers of 1967, 1968, and 1969, stump sprouts in the phase one clearcut area were controlled with 2, 4, 5-T and 2, 4-D herbicides, applied by back-pack mistblowers [Lynch et al., 1975]. In June, 1974, an aerial application of 2, 4, 5-T and atrazine was applied to both the phase one and phase two clearcut areas. The purpose of the herbiciding was to deaden both woody and herbaceous vegetation as part of a water-yield experiment.

The treated watershed is adjacent to two control watersheds 303 and 257 acres in size. Soils, vegetation, and geology are similar on all watersheds. Most of the soils are residual, having developed in place through the weathering of the underlying strata. The Rose Hill shale formation underlies most of the lower slopes. The upper part of the lower slope, the middle slope, and most of the upper slope are underlain by Castanea sandstone. Tuscarora quartzite underlies the ridge tops.

Weekly sampling for nutrient concentrations in streamflow began in 1972. Therefore, a direct comparison between the clearcut and control watersheds is necessary to evaluate treatment effects. Clearcutting had little, if any apparent effect on the concentrations of selected elements the first year after completion of the phase two clearcut [Lynch et al., 1975]. During the second year after cutting, there was a slight decrease in the calcium and sulfate concentrations and a slight increase in the nitrate-nitrogen concentrations [Table 1]. Average nitrate-nitrogen during the period October 1973-May 1974 was 0.10 ppm. There were little or no changes in the concentrations of magnesium, potassium, and sodium.

After all vegetation was deadened in June, 1974, there were substantial changes in nutrient concentrations on the treated watershed. There was a further reduction in calcium and sulfate concentrations and a larger increase in nitrate-nitrogen [Table 2]. There was an indication of a slight rise in potassium concentration, while magnesium and sodium concentrations did not seem affected by the treatment. Average nitrate-nitrogen concentrations increased to 2.08 ppm during the seven-month period after herbiciding. The nitrate-nitrogen concentration remained below 0.4 ppm for the first three months after herbiciding and then increased sharply from September through December. The maximum average monthly concentration was 5.0 ppm. The maximum concentration measured was 8.4 ppm.

Concentrations of nutrients in streamflow on both the clearcut and control watersheds tend to increase from the headwaters to the mouth of the watershed. This reflects, in part, the geologic formations over which the streams flow and opportunities for soil-water leaching. Except for nitrate-nitrogen, there was no evidence of significant changes in nutrient concentrations in the upper half of the stream draining the clearcut watershed during October 1973-May 1974. Nitrate-nitrogen increased from 0.04 ppm at the headwaters of the clearcut watershed to 0.15 ppm at a point on the stream one-half the distance between the headwaters and the mouth of the watershed.

TABLE 2

Average Nutrient Concentration of Stream Water
On Partially Clearcut and Control Leading Ridge Watersheds
June - December 1974.

ELEMENT	CONTROL LR-1	TREATED[1/] LR-2	CONTROL LR-3
	-------------	PPM	-------------
CALCIUM	5.38	2.32	6.28
MAGNESIUM	2.88	2.06	2.30
POTASSIUM	1.08	1.58	1.13
SODIUM	1.12	0.96	1.16
SULFATE	8.20	4.68	7.23
NITRATE - N	0.04	2.08	0.08

[1/] All vegetation on clearcut portion of watershed was deadened with herbicides (2, 4, 5-T and atrazine) June 1974.

After deadening all vegetation in June, 1974, with herbicides, there were small increases in potassium and nitrate-nitrogen at the headwaters site (located in the upper portion of the phase-2 clearcut). These increases were larger farther downstream, especially in the middle portion of the clearcut at the one-third and one-half distance sampling points [Table 3].

EFFECTS OF VEGETATION MANAGEMENT ON STREAM WATER TEMPERATURE

The effect of timber-harvesting practices on water temperature is related directly to the surface area of stream exposed to direct sunlight. This is due primarily to the energy increase created when the low-intensity, diffused light under the forest is changed by canopy removal to direct solar radiation [Levno and Rothacher, 1967]. Before 1950, the effect of clearcutting on the water temperature of small mountain streams had not been intensively studied. Since then, a number of studies have shown a substantial change in water temperature of a stream after the surrounding vegetation has been cut.

Impacts

Changes in streamwater temperature can affect fish by influencing their metabolic rate, changing the oxygen content of water (dissolved oxygen in water is inversely proportional to water temperature), influencing hatching and development, and affecting migration. Reduced growth, vigor, and resistance to disease probably are the main effects of high temperatures [Fredriksen, 1972a]. Populations of most of the fish pathogens greatly increase as water becomes warmer. Ideal water temperature for cold-water fish is between 50 and 65°F; generally they cannot tolerate temperatures over 75°F [Trippensee, 1953].

Water temperature may also affect taste, odor, and color of streamwater. Under some conditions, temperature increases stimulate excessive production of algae, thus damaging the quality of water for human consumption, depleting the oxygen supply for aquatic organisms, and lowering the aesthetic quality of streams [Brown et al., 1971].

Timber Harvesting Effects

On the Leading Ridge Watershed in central Pennsylvania, weekly water temperatures were monitored after the clearcutting of 21.3 acres on the lower slope of a 106-acre watershed. During the first year, summer water temperatures on the treated watershed were considerably higher than those on the control watershed. Mean water temperatures increased 4°F from (58° to 62°F) during the same period. Minimum weekly water temperatures during the summer months were generally unchanged as a result of clearcutting [Lynch et al., 1975]. It should be noted that this 21-acre clearcut essentially represents a riparian cut, because all the main stream channels on the watershed are located within its boundaries. Furthermore, the regrowth of the woody vegetation was prevented by the application of herbicides.

Five years later, an additonal 27 acres on the middle slope of the same watershed were clearcut. The combined effects of both clearcuts on stream temperatures were essentially the same as those observed following the lower-slope clearcut. During the first year after cutting, average summer maximum weekly water temperature on the treated watershed was 8°F higher (66 to 74°F) than on the control watershed. The maximum monthly average water temperature of 77°F was recorded in July, an 11°F increase over the average July maximum water temperature on the control watershed. Mean summer weekly water temperature was 4°F higher (61 to 65°F) on the treated area; minimum weekly water temperatures remained unchanged [Lynch et al., 1975].

After all vegetation on both the lower and middle-slope clearcuts was deadened by herbicides in June, 1974, additional increases in streamwater temperatures were measured. The maximum daily water temperature on the treated watershed during July 8-28, 1974, averaged 78°F and ranged from 67° to 82°F [Table 4]. This represents a 14°F increase when compared with the 64°F average maximum daily stream temperature on the control watershed. Control watershed maximums ranged from 62 to 68°F. Although daily maximum water temperature on the control watershed never exceeded 68°F, stream temperatures on the herbicided area exceeded 75°F on 18 days of this 21-day period.

TABLE 3

Average Nutrient Concentration on Stream Water On Partially Clearcut Leading Ridge Watershed No. 2 June 1974 - Dec. 1975

ELEMENT	Sample Site Location 1/ HEADWATER	1/3	1/2	WEIR
	----------	----------	PPM	----------
CALCIUM	0.63	1.12	1.55	2.32
MAGNESIUM	0.76	1.27	1.69	2.06
POTASSIUM	0.93	1.23	1.45	1.58
SODIUM	0.41	0.60	0.74	0.96
SULFATE	3.49	2.68	3.40	4.68
NITRATE - N	0.24	1.92	2.40	2.08

1/ Fractions indicate distance sampling site is between headwater of stream (near top of phase 2 clearcut) and weir.

TABLE 4

Maximum Daily Stream Water Temperatures on Control (LR-1) and Treated (LR-2) Watersheds During July 8-28, 1974.

WATERSHED	MAXIMUM DAILY WATER TEMPERATURE AVERAGE	RANGE
	°F	°F
CLEARCUT	78	(67-82) 1/
CONTROL	64	(62-68)
DIFFERENCE	+14	-----

1/ On 18 of the 21 days measured, water temperature on the treated watershed exceeded 75°F for at least 15 minutes.

Minimum daily stream temperatures were not greatly changed by the herbicide treatment [Table 5]. Minimum daily stream temperatures during July 8-28, 1974, averaged 3°F higher (59 to 62°F) on the treated area when compared with the control watershed. Minimum daily water temperatures ranged from 58 to 67°F on the herbicided area as compared with 55 to 62°F on the control watershed.

The range in average daily stream temperature fluctuations increased substantially from the headwaters to the mouth of the watershed [Table 6]. The perennial stream channel emerges in the upper portion of the clearcut area, where the average daily water temperature fluctuation during July 8-28, 1974, was only 1°F. At a point one-third of the distance from the headwater site to the stream-gauging station, the average daily water temperature fluctuation increased to 3°F, and at the two-thirds point to 7°F. This was still less than half the average daily range of 16°F measured just above the stream-gauging station.

The average daily maximum water temperature changed from 57°F at the headwaters to 78°F near the stream-gauging station, an increase of 21°F along the 2,600-foot section of stream. The average daily minimum temperature changed from 56 to 62°F over the same distance, an increase of 6°F. However, the average daily minimum temperature did not change in the lower two-thirds of the stream, while the average daily maximum stream temperature increased continuously from the headwaters to the mouth of the watershed [Table 6].

On the Fernow Experimental Forest near Parsons, West Virginia, the average maximum stream temperature during the first two growing seasons after 74 acres of an east-facing watershed was commercially clearcut was 8°F higher (59 to 67°F) than on a control watershed. The highest water temperature recorded on the clearcut watershed was 79°F; however, numerous temperatures above 75°F were observed [Eschner and Larmoyeux, 1963]. As a result of regrowth, stream temperatures returned to precutting levels within five years after cutting [Kochenderfer and Aubertin, 1975].

Maximum growing-season stream temperatures for a south-facing watershed (54 acres), on which the lower half was clearcut and herbicided, averaged 10°F higher than the control watershed. On an east-facing watershed (59 acres), where the clearcut operation on the lower half left the stream partly covered by slash that shaded the channel, the maximum growing-season stream temperatures averaged 4°F higher than the control. Deforesting/herbiciding the upper halves of these watersheds had no appreciable effect on stream temperatures. However, no additional stream channels were exposed to direct radiation on the deforested upper halves [Kochenderfer and Aubertin, 1975].

There is an indication that clearcutting increased stream temperature fluctuations and lowered minimum temperatures during the dormant season. The reason is unknown, but may be associated with increased wind movement through the clearcut area [Aubertin and Patric, 1974].

On the Hubbard Brook Experimental Watersheds in New Hampshire, Likens et al. [1970] noted higher stream temperatures on a completely deforested watershed than on the forested control watershed. Maximum growing-season stream temperature on the treated area was 71°F, while the control stream never exceeded 59°F.

TABLE 5

Minimum Daily Stream Water Temperature On Control (LR-1), and Treated (LR-2) Watersheds During July 8-28, 1974.

WATERSHED	MINIMUM DAILY WATER TEMPERATURE AVERAGE	RANGE
	°F	°F
CLEARCUT	62	(58-67)
CONTROL	59	(55-62)
DIFFERENCE	+ 3	----

TABLE 6

Fluctuation In Average Daily Water Temperature At Various Points Along Stream Draining Clearcut Portion of Leading Ridge Watershed 2 During July 8-28, 1975[1]

LOCATION	HEADWATER	1/3 [2]	2/3 [2]	Weir
AVG. DAILY MAX.	57	65	70	78
AVG. DAILY MIN.	56	62	63	62
AVG. DAILY RANGE	1	3	7	16

[1] One month after application of herbicides.

[2] Fractions indicate distance sampling site is between headwater of stream (near top of phase 2 clearcut) and weir.

At the Coweeta Hydrologic Laboratory in North Carolina, stream temperature increases as high as 20°F (65 to 85°F) were observed on a watershed after clearcutting and conversion of a deciduous forest to farm crops. However, with a less severe treatment in which the cove timber was deadened on another watershed, summer maximum stream temperatures increased only 4 to 5°F. The highest maximum stream temperature recorded was 70°F. After the deadened cove timber was felled and the rest of the watershed was cleared, summer maximum stream temperatures increased 5 to 6°F, only 1°F higher than the cove treatment. Highest stream temperature recorded was 75°F. Temperatures returned to pretreatment levels in the fourth year after cutting [Swift and Messer, 1971].

The importance of overhead shade has also been demonstrated by Swift and Messer [1971]. In studies in North Carolina, they found that in clearcut areas where streambank vegetation was uncut, summer maximum water temperatures remained unchanged when compared with a control watershed. Where streambank vegetation had regrown after a clearcutting eight years before, summer maximum water temperatures were even lower than those under uncut mature hardwood forests. This was attributed to the dense shading caused by sprout growth that developed after cutting. They concluded that the magnitude of water temperature increase depends upon the amount of vegetation (shade) removed along the stream.

Another such study, conducted in Oregon and reported by Brown and Krygier [1967], showed that water temperature increases of 16°F (52 to 68°F) occurred as early as May when the stream passed through a 1,300-foot clearcut stretch. However, after the stream had passed under 700 feet of undisturbed canopy, maximum water temperatures were reduced as much 8°F, indicating considerable cooling in the shaded stretch. However, Brown, et al., [1971] later concluded that "shaded reaches downstream from a clearcut cannot be relied on to cool heated streams. Cooling that does occur can often be attributed primarily to inflow and mixing of cooler ground water."

Controlling Stream Temperatures on Timber Harvesting Sites

Recently the concept of buffer strips has been getting a great deal of attention as a method of protecting the stream ecosystem in timber-harvesting operations. The idea is to leave a strip of relatively undisturbed vegetation along the banks of streams at the time adjacent lands are logged so the stream will be buffered from such adverse effects as siltation, slash and debris accumulation, and temperature increases.

Bengeyfield [1973], while studying streams in West Virginia, found that weekly maximum stream temperatures on both a control watershed and a clearcut watershed with a buffer zone were essentially the same during the summer months. The average difference between them was 0.8°F, the control watershed slightly cooler. During the same period, maximum temperatures on a clearcut watershed that had no buffer zone averaged 11°F higher than the control. The highest maximum stream temperature was 77.5°F.

The previously mentioned strip-cutting in New Hampshire also utilized a buffer zone to provide shade and protect the stream environment. Summer stream temperatures increased less than 2°F after the cutting [Hornbeck and Federer, 1975].

Brazier and Brown [1973] found that commercial timber volume alone is not an important criterion for temperature control. For the streams studied, they found that the maximum shading ability of the average strip was reached within a width of 80 feet; 90 percent of that maximum was reached within 55 feet. Specifying standard 100 to 200-foot-wide buffer strips for all streams, which usually assures protection, generally will include more timber in the strip than is necessary.

The above studies illustrate the effect that maximum clearance of vegetation has on summer temperatures of small streams. However, they also indicate that temperature increases are not uniform along the entire length of exposed stream. In addition, when only a portion of a drainage basin is clearcut, or when adequate buffer zones are employed, these adverse effects are minimized.

EFFECTS OF VEGETATION MANAGEMENT ON STREAM TURBIDITY AND SEDIMENTATION

The effects of forest treatments on the sediment load carried by streams varies considerably. Some forest areas can undergo severe treatment and remain relatively stable. Others erode severely after only slight disturbance. Such differences in the hydrologic behavior and stability characteristics of forest land can usually be traced to variations in climate, topography, geology, and soils. These variations tend to be balanced by vegetation in ways that promote soil stability in most forest regions. However, a significant portion of the nutrients carried from the land to water may be attached to eroded soil particles and organic particulate matter [Fredricksen, 1972b].

Timber cutting itself--unconfounded by skidding and road-building disturbances--has little if any effect on the turbidity of streams [Bates and Henry, 1928; Hoover, 1944; Leiberman and Hoover, 1951; and Likens et al., 1970]. However, when the effects of timber cutting are combined with effects caused by improper logging or road construction, the quality of water deteriorates. Numerous studies throughout the United States have emphasized the seriousness of this problem. The results of these studies have been summarized by Packer [1967].

Impacts

Accelerated erosion and sedimentation rates can lead to many environmental consequences. Turbid water blocks light transmission and reduces primary production of aquatic plants. Fine sediment that fills the interstices of gravel in spawning beds reduces the circulation of water and the oxygen supply necessary for developing fish embryos [Fredrickson, 1972a]. It can cause inflammation of the gill membranes and eventual death of the young fish. In addition, organic sediments require oxygen for decomposition, which reduces that available for fish respiration. Other problems associated with increased sedimentation include reduction in reservoir storage and stream channel water-carrying capacities, water-quality impairment and health hazards, increased cost of water treatment, and a reduction in hydrologic amenities.

Timber Harvesting Effects

Reihhart et al. [1963] observed serious pollution when "logger's choice" skid-

TABLE 7

Streamflow Turbidity Values for Leading Ridge Watersheds

TREATMENT AND PERIOD	STORM TURBIDITY AVG.	MAX.	NON-STORM TURBIDITY AVG.	MAX.
		Jackson Turbidity Units		
LOWER SLOPE CLEARCUT				
DURING LOGGING	196	550	---	---
AFTER LOGGING	9	119	2	24
UNDISTURBED CONTROL	3	29	1	17
MIDDLE SLOPE CLEARCUT				
DURING LOGGING	---	---	2	11
AFTER LOGGING	4	13	1	6
UNDISTURBED CONTROL	2	7	2	7

roads, with little if any drainage facilities, were used on a commercial clearcut and a diameter-limit cut on the Fernow Experimental Forest. Maximum turbidities were 56,000 and 5,200 JTU (Jackson Turbidity Units) respectively. On extensive and intensive selection cuts, with carefully constructed skidroads and water bars, maximum turbidity reached only 210 and 25 JTU, respectively, during cutting and subsided almost immediately after logging ceased. Maximum turbidity on the control watershed never exceeded 15 JTU. Regardless of the logging practice employed, the impact of cutting on stream turbidity was greatest during and immediately after logging.

On another Fernow clearcut watershed, logged with a carefully planned and maintained road system and streamside buffer zone (33 to 66 feet wide along each side of the stream), only minor increases in water turbidity occurred. Road gradients were generally restricted to 10 percent. Erosion-control measures such as outsloping, broadbase dips, culverts, and gravel were used as needed. Large trees in the buffer strip were selectively harvested by winching logs up to the haul road. After completion of operations, the logging roads were seeded to grass. During the logging period, nonstorm turbidity averaged 6 JTU with a maximum of 90 JTU during stormflow, except for one incident when erosion-control structures were damaged during a severe thunderstorm, resulting in stream turbidity of 550 JTU. The first year after logging, the corresponding turbidity values were 5 JTU and 35 JTU [Aubertin and Patric, 1974].

The highest turbidity recorded for six storms on the Leading Ridge Watershed 21-acre clearcut in central Pennsylvania was 550 JTU during logging [Table 7]. The next highest turbidity recorded was 129 JTU. Increased stream turbidity, which could be traced to scarified log-loading areas, decreased immediately after completion of logging. Average storm turbidity from the clearcut decreased from 196 JTU during logging to 11 JTU the first year after logging, and ranged from 3 to 13 JTU during the next four post-treatment years. On the control watershed, storm turbidity never exceeded 29 JTU and consistently averaged less than 5 JTU during the study [Lynch et al., 1975].

On this clearcut, precautions were taken to reduce the sediment load resulting from logging operations. These precautions included restricting cutting to periods in the winter when the ground was frozen, careful location and construction of logging roads and skidtrails, and seeding and fertilizing of road areas and log-loading areas to facilitate rapid revegetation.

An insufficient number of storms occurred during the actual logging of the middle-slope timber harvest on the Leading Ridge Watershed to determine its effects on storm turbidity. However, non-storm turbidity measurements indicated that the effects of the middle-slope cutting on stream turbidity were considerably less than those from the lower-slope cutting [Table 7]. Non-storm turbidity measurements made during this period averaged only 2 JTU on both the treated and control watersheds. Maximum non-storm turbidity measured was 11 JTU on the treated watershed and 7 JTU on the control watershed.

Storm turbidity samples taken the first year after completion of the middle-slope logging operation averaged 4 JTU on the treated watershed and 2 JTU on the control watershed. Maximum storm turbidity during this period was 13 JTU on the treated area and 7 JTU on the control. Non-storm turbidity the first year after completion of logging never exceeded 6 JTU and averaged only 1 JTU on both the treated and control watersheds.

On the Hubbard Brook, N.H., strip-cut watershed referred to earlier, stream turbidity was kept to a minimum by following known logging precautions. In the two-year period during and after cutting the strips, only 6 of 147 stream samples had turbidity values exceeding the drinking water standard of 5 units. The maximum value recorded was 38 JTU [Hornbeck and Federer, 1975].

In general, bare soil exposed by road-building, and to a much lesser extent by log landings, has long been recognized as the major source of stream sediments associated with logging operations. Careless logging can cause very turbid water, while logging on carefully located, constructed, and maintained road systems and skid trails results in only minor increases in turbidity. [Hornbeck, 1968].

DISCUSSION

Nutrient concentration data alone, without information on the volume of streamflow, can be misleading. Increases in water yield from a clearcut watershed may more than compensate for decreases in concentrations when total amounts of nutrients are considered. However, major changes in nutrient leaching due to vegetation removal can be discerned with concentration data.

In general, except for studies in New Hampshire, there have been only slight and somewhat temporary changes in nutrient concentrations due to normal timber harvesting operations. When all vegetation has been deadened by herbicides, significant nutrient leaching occurs. This points out the importance of establishing vegetative cover on cutover areas as soon as possible to hold the nutrients on site.

Clearcutting temporarily stops the normal circulation of nutrients between the plant cover and soil system until the area is at least partially revegetated. Maximum increases in mean temperatures and available moisture can be expected to favor rapid decomposition of organic matter and increase nutrient release to the soil. Many forest soils have a high capacity to retain these nutrients against loss to ground water or surface streams [Moore and Norris, 1974]; others, however, do not [Reinhart, 1973].

Since most of the nitrogen in the humus layer of an uncut forest is not readily available to plants, the amount flushed from the organic matter as nitrate does not necessarily represent a loss to plant growth. In fact, clearcutting probably increases the amount of available nitrogen by increasing the rate of biological nitrification. This may explain in part the rapid regrowth that usually follows clearcutting [Sartz, 1975]. Such rapid growth following disturbance tends to minimize losses of nutrients from the ecosystem, thus promoting a return to steady-state cycling characteristic of a forested watershed [Marks and Borman, 1972]. Partial cutting by selection or shelterwood harvesting systems, strip-cutting, and various thinning and pruning operations have less effect on nutrient cycling, depending upon the amount of material returned to the forest floor.

Careless logging can cause very turbid water, while logging on carefully located, constructed, and maintained road systems and skid trails results in only minor increases in turbidity. The use of buffer strips to protect streams, where only selection cutting is allowed, will aid in minimizing any adverse effects of sedimentation, nutrient leaching,and stream temperature increases.

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EVALUATION OF SEDIMENT YIELD
DUE TO HOUSING CONSTRUCTION:
A CASE STUDY

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ABSTRACT

This paper reports findings of a research project to determine sediment yields in the area of Cedar Hill, Virginia, near Virginia Beach. Soil-erosion rates at construction, semi-construction, and well-developed sites in this area were estimated using a modified Universal Soil Loss Equation. Sediment yields at the three sites and at a downstream monitoring station then were measured and compared with the estimates. Dilution factors were studied as a function of rainfall intensities, which are believed to correlate with the characteristics of the area undergoing construction. The dilution factor was defined as the ratio of suspended-sediment concentration on the site as compared to that for an off-site specific downstream station at the peak hours of runoff hydrographs at that location.

INTRODUCTION

The tremendous volume of urban-construction erosion and its impact on society has become a major cause for public concern. Soil sediments are considered non-point pollutants if they interfere with the use of water for domestic use, navigation, drainage, recreation, agriculture, or biological or ecological functions. Sediments, including nutrients and pesticides in sediments, influence water quality, and thereby affect the growth of organisms in lakes, rivers, estuaries, and the marine environment. Public health may be affected in a number of ways. Economic loss may result from actions such as the closing of oyster beds due to high concentrations of sediment. Unless urban construction and development are well planned and carefully managed, the total environment will be degraded as the result of urban sediment erosion, transport, and deposition.

It is desirable that planners and designers understand the nature of the problem in general, the mechanics of soil erosion and sediment yield, the availability of erosion-control guidelines and technology, and legislative measures such as erosion and sediment control laws and local ordinances. Numerous publications are available [Environmental Protection Agency, 1973; Guy, H. P., 1970; Guy, H. P., and Jones, D. E., Jr., 1972; Guy, H. P., 1974; Guy, H. P., 1975a; Guy, H. P., 1975b; Heinemann, H. G., and Piest, R. P., 1975] in the general areas, and these

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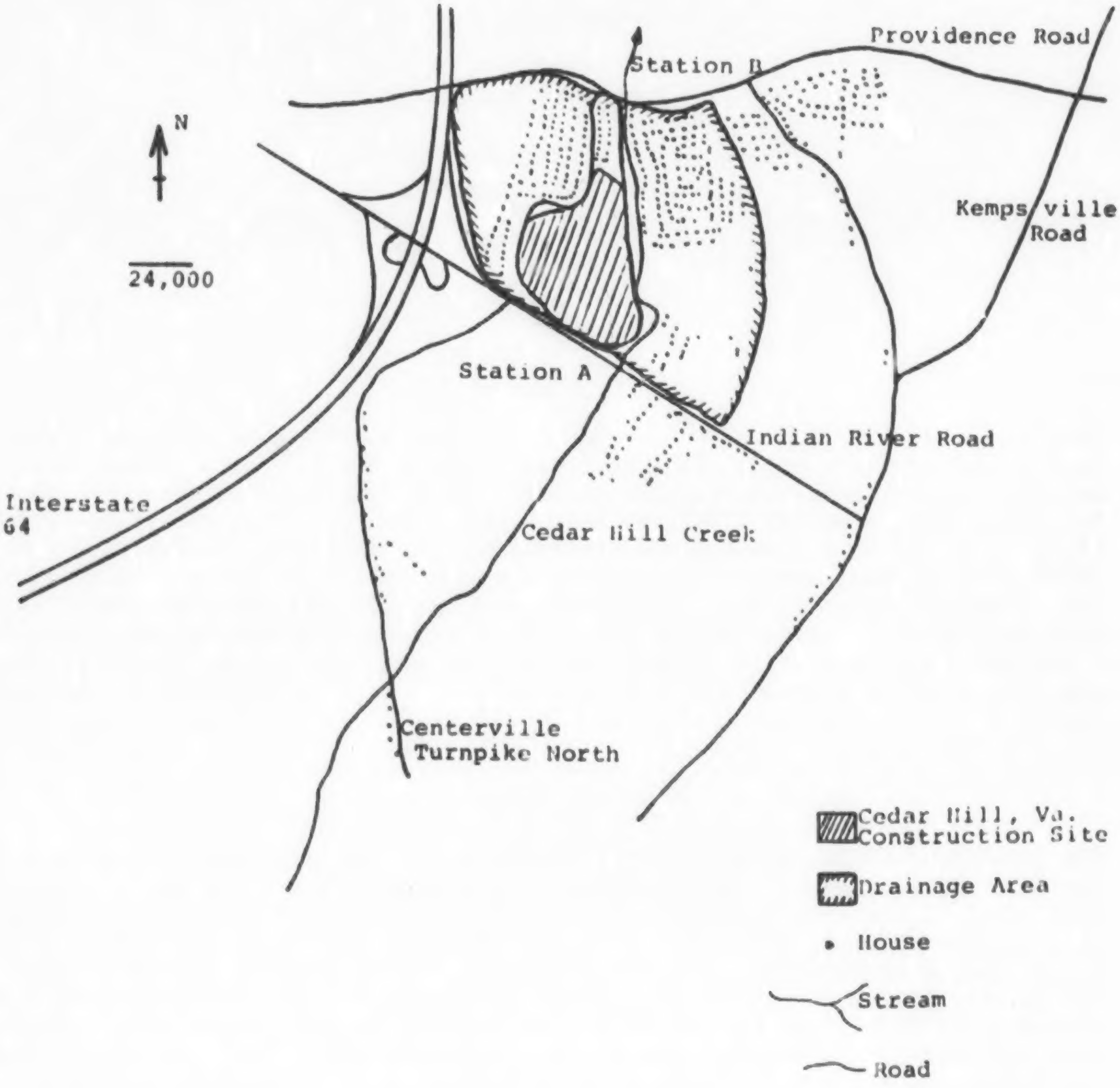
FIGURE 1

Drainage Area Under Study


N
24,000
Providence Road
Station B
Kempsville Road
Station A
Indian River Road
Interstate 64
Cedar Hill Creek
Centerville Turnpike North
Cedar Hill, Va. Construction Site
Drainage Area
House
Stream
Road

have led to more specific literature. It is beyond the scope of this paper to review the entire urban sediment problem. This paper presents some preliminary findings of an ongoing project of evaluation of sediment yield in a housing construction area of Cedar Hill subdivision, Virginia Beach, Virginia.

The development of Virginia Beach differs from other Tidewater Virginia cities. The city's rapid growth is primarily due to activities related to beach resorts, seaports, naval bases, large industries, and commercial enterprises. Urban construction, involving the change from farm lands to residential areas, has accelerated in recent years. It is desirable to insure acceptable water quality as one aspect of coastal resources management by minimizing the impact of the urban sediment problem.

Although the city has an erosion and sediment control ordinance, its effective implementation requires better understanding of the erosion, transport, and deposition processes in this part of the coastal plain. Practically no streamflow-gaging or sediment measurement stations have been established in this area, since most of the creeks are in the tidal region and non-tidal creeks are small and short in length. The applicability of the Universal Soil Loss Equation for either rural areas or its slightly modified form for construction areas needs to be studied in the context of the flat topography.

THE WATERSHED

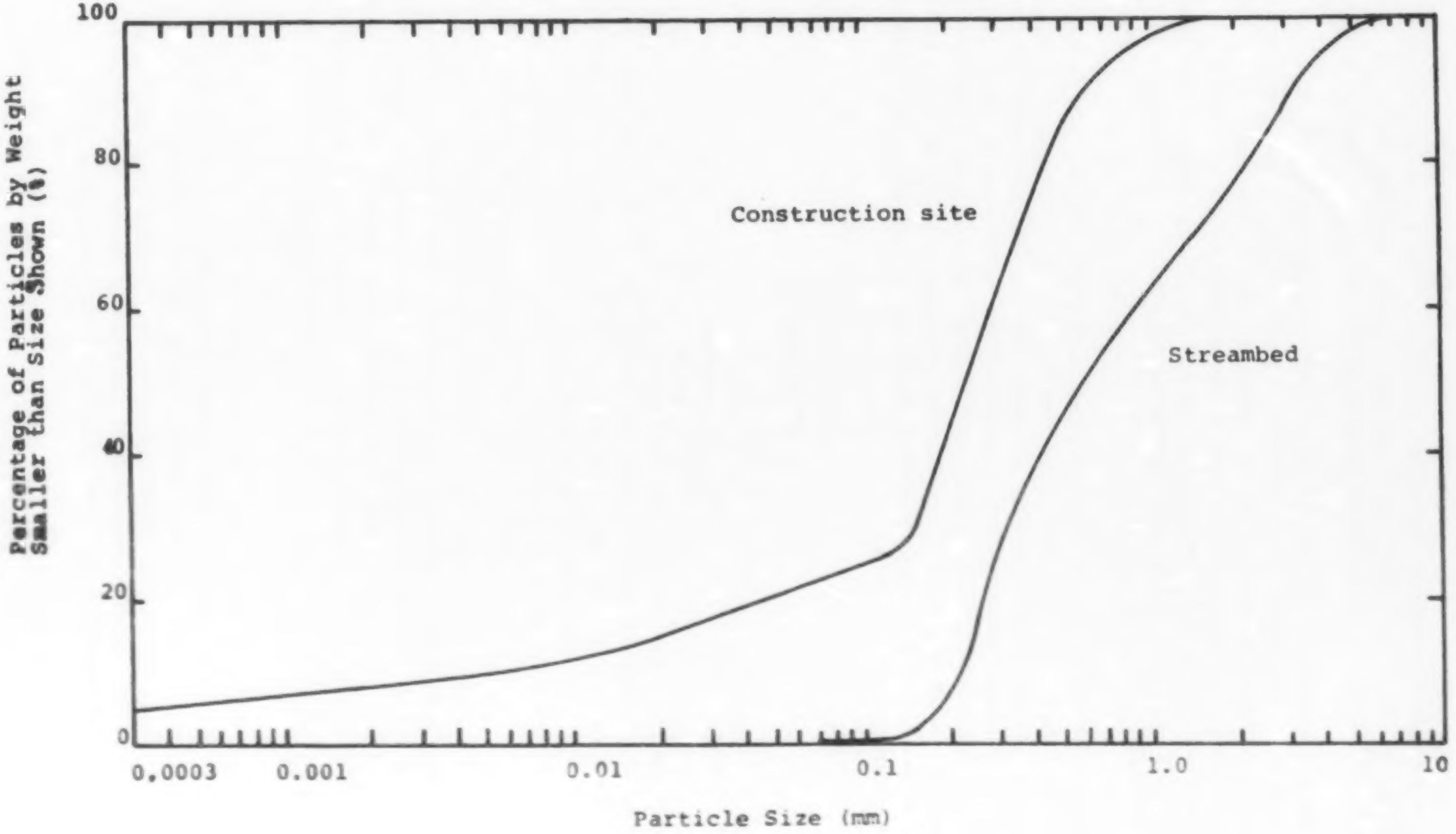
Figure 1 shows the location of the study area. Because of the flat terrain in the coastal plain, watershed boundaries normally are difficult to identify. When they are located, it is likely that the area is too large for an efficient sediment-yield study because of the relatively small size of construction sites. The drainage area upstream of gaging Station A [Figure 1] is relatively big and flat, with significant coverage by small swamps, storage ponds, and tall vegetation. Consequently, the time of concentration for surface runoff at Station A is long, the runoff volumes and peak discharges are small, and the suspended sediment concentrations are low. On the other hand, the drainage area between Stations A and B is well developed, with Cedar Hill the only site subjected to the severe soil disturbances of housing construction. At Station B, the time of concentration, runoff volume, and suspended-sediment concentration are quite different from Station A. These features suggest selection of Point A as the reference station (rural conditions) and Point B as the monitoring station for the runoff and sediment measurements of the residential and construction areas. Cedar Hill Creek, which runs through the area, is a non-tidal stream with an average width of 20 feet, embankment height of 6 feet, and slope of 0.0033 for this reach.

The drainage area between the two stations is 290 acres. Cedar Hill subdivision has an area of 97.5 acres. The surface coverage within the drainage area is classified as follows: 7.44 percent woods, 3.06 percent pond, 1.74 percent totally exposed area, 23.32 percent impervious surface including roofs, driveways, and streets, and 64.42 percent semi-exposed area including lawns, gardens, farm land, and stripped but unconstructed land with some vegetation growth.

The land use of the construction site formerly was agricultural. The subdivision was laid out in 1971 and house construction began late in that year. The neighboring subdivision inside the sub-drainage area was developed in the mid 1960's. Generally

FIGURE 2

Particle Size Distributions for Soils at Streambed and Construction Site



only four to five single-family houses are under construction at a given time. Hence the exposed area undergoing active construction is relatively constant at any time. The building permit was issued before approval of the city erosion and sediment ordinance, so no sediment-control measures were taken. Most of the eroded sediments were carried to the creek through the storm drainage system via street-drain inlets.

SOIL AND SEDIMENT INFORMATION

Soil types were obtained from the local Soil Conservation Service. The soils inside the drainage area studied fall into two major categories. One is Woodtown, a moderately well-drained soil with fine sandy loam and loamy sand substata, a high seasonal water table, permeability of 2.0-6.0 inches/hour, and available moisture capacity of 0.14-0.19 inch per inch of depth. The other type of soil is Othello, which is poorly drained, has a high seasonal water table, fine sandy loam and loose fine sand, permeability of 0.63-2.0 inches/hour, and available moisture capacity of 0.16-0.23 inch per inch of depth. Othello is the soil at the construction site. Particle size distributions for this soil and the stream bed material are shown in Figure 2. Only 0.5 percent of the riverbed sediment is in the range of silt and clay sizes (less than 0.073mm), whereas 23 percent of the soil at the construction site falls in this range. Nearly all the suspended sediment moving at the monitoring station downstream from the construction site was of silt and clay sizes. Thus, the sediment sampled was primarily washload from land erosion, not channel erosion.

Analysis of suspended-load, channel-bed, and construction-site materials for grain size were made following standard methods specified by the American Society for Testing Materials (ASTM).

DATA COLLECTION

Water and sediment-discharge measurements were made at Gauging Stations A and B [Figure 1], upstream and downstream from the drainage area under study. Stream stages were continuously recorded by means of Stevens A-35 recorders. Streamflow ratings were established using a Pygmy Type F-583 water-current meter from Weather Measure Corporation. Suspended-sediment samples were collected at appropriate time intervals at both stations during storm periods using a USDH-48 sampler. Concentrations of suspended sediment were determined using the evaporation method specified in U.S. Geological Survey series on techniques for water resources investigations. Type P501-I remote recording rain gauge with Type P521 event recorder, both from Weather Measure Corporation, were installed for rainfall data collection at the midpoint between Stations A and B, near the center of the drainage area. Suspended-sediment samples also were collected at the street-drain inlets draining well-developed areas, construction sites, and semi-construction areas where the land was stripped but unconstructed. Concentration data at the street-drain inlets were obtained (1) to define the variation of sediment concentration with respect to time at the three sites, and (2) to study the effect of dilution. Samples also were collected for these purposes at the peak hour of river stage at the downstream station.

FIGURE 3

Data for Streamflow and Suspended Sediment Concentration due to Storm March 30, 1975

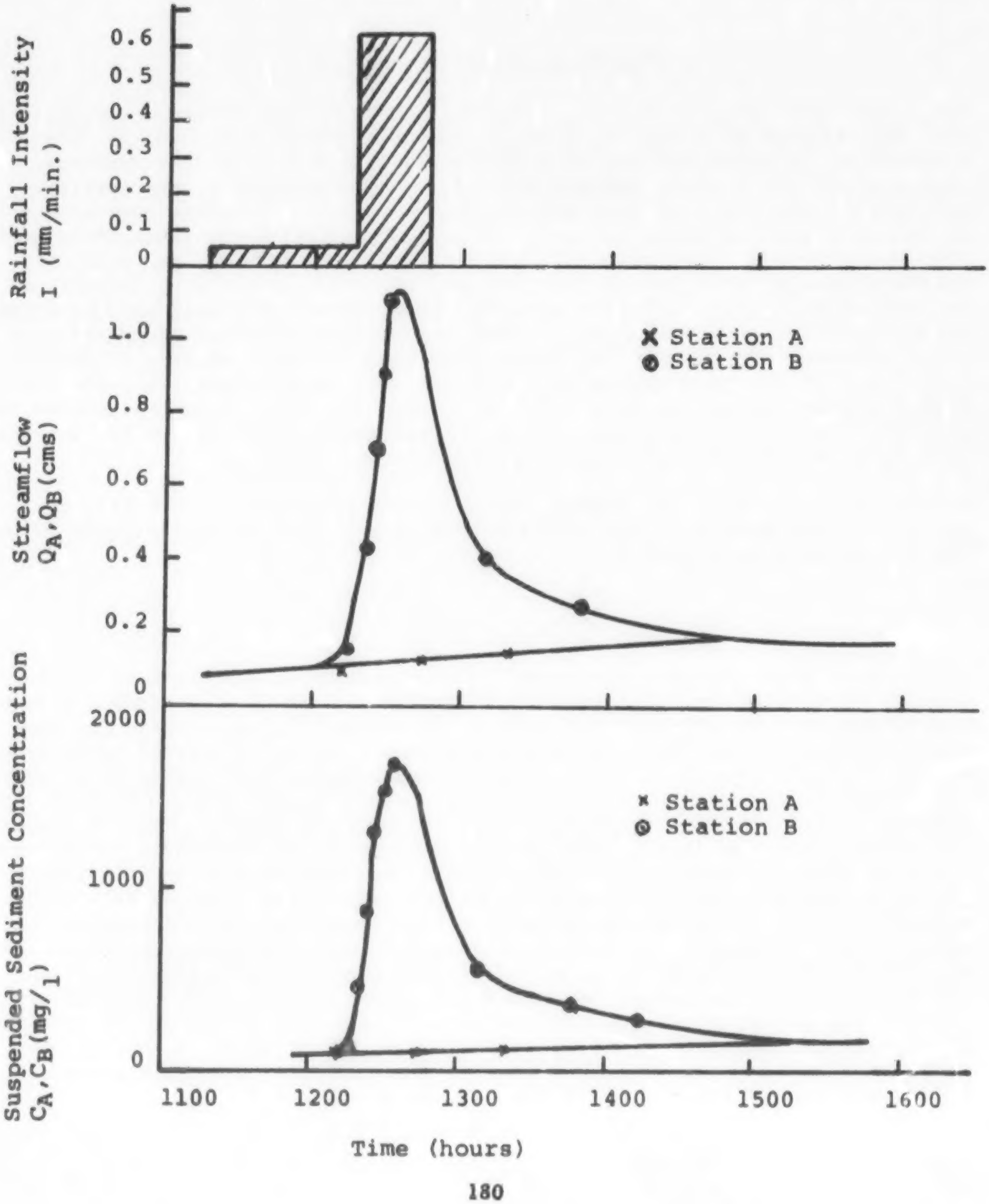


FIGURE 4

Net Surface Runoff and Sediment Contributed from the Drainage Area Undergoing Construction for Storm March 30, 1975

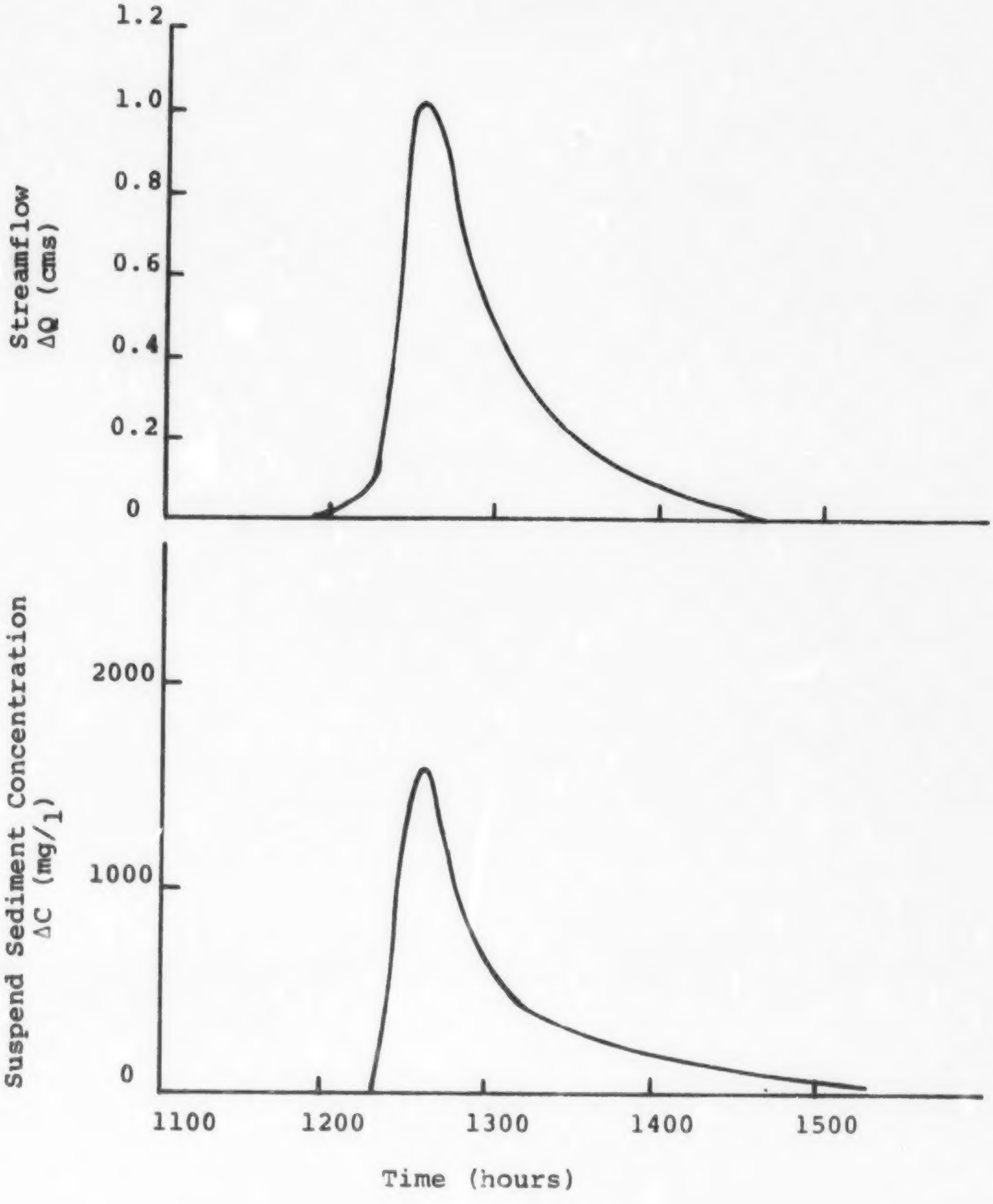
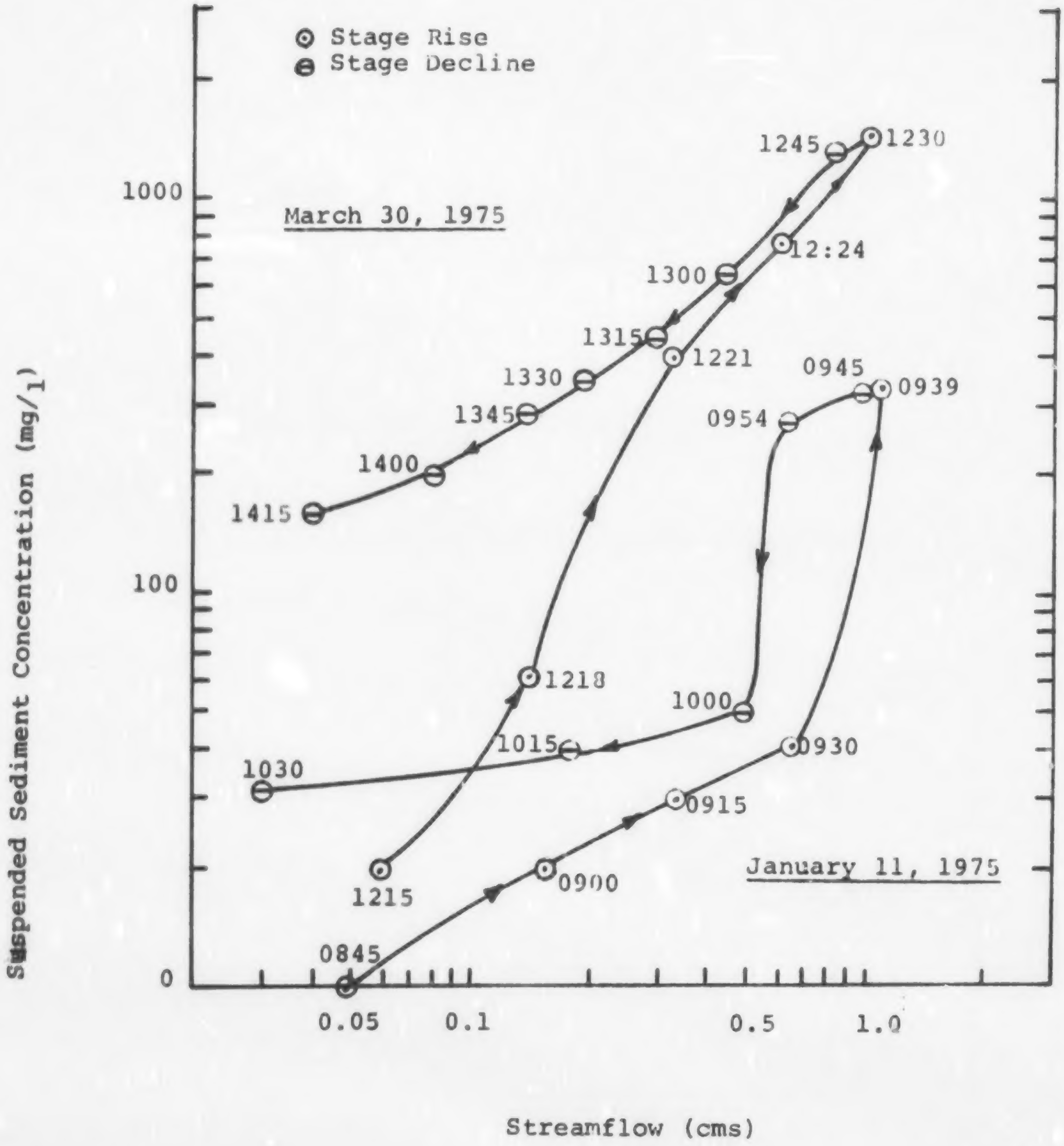


FIGURE 5

Sediment Concentration/Streamflow Rating
for Storms January 11 and March 30, 1975



RESULTS AND ANALYSIS

Typical data obtained can be seen from the results of a particular storm on March 30, 1975. Suspended-sediment concentrations and discharges upstream (Station A) and downstream (Station B) as a function of time are plotted for this storm in Figure 3, with rainfall data inserted for reference. The extremely low and stable streamflow and suspended-sediment concentration at Station A, the upstream reference station, verifies the observation previously made on the characteristics of the drainage area upstream from this point.

The response of Station B to the surface runoff and sediment load from the drainage area undergoing construction is dramatic. The net suspended-sediment concentration and discharge between the two stations represent contributions from the drainage area under consideration. They are plotted in Figure 4 as a function of time. The peak time for the runoff hydrograph is a reflection of the rainfall pattern and reveals that the time of concentration for the drainage area between Stations A and B is very short. Rainfall data are used for the calculation of erosive energy, whereas the hydrograph and sedimentgraph are used for the computation of water and sediment yield. Both results are used in conjunction with the estimation of soil erosion rate and sediment-delivery ratio. Curves for the sediment concentration/streamflow rating for storms of January 11 and March 30, 1975 are shown in Figure 5.

During the fall and early winter of 1974, the weather was extremely dry with occasional light showers and drizzles. Figure 6 indicates a very small difference in discharge and sediment concentration between the two stations corresponding to light intensity and long-duration rainfall exemplified by the rain of January 4, 1975. The amount of sediment washed away from the construction land is relatively insignificant and unmeasurable.

For a rainfall of 16 minutes duration and 1.06 in/hr uniform intensity, the sediment concentration for the samples collected at street drain inlets is plotted in Figure 7 as a function of time. This shows the on-site erosive phenomena--how the soil is eroded and transported along the gutter to the street-drain inlet. C-1 in Figure 7 is for a well-developed area. C-2 is on the construction site. C-3 is in a semi-construction area, and C-4 is at a location between C-2 and C-3. The magnitude of sediment concentration at each location is relatively high at the beginning of the rain and decreases thereafter. The time rates of change of sediment concentration are shown in Figure 8 for each case. They are almost the same, except for C-1. The average sediment concentration occurs at approximately the same time when the runoff hydrograph at the downstream station reaches its peak from the runoff between Stations A and B. This corresponds approximately to the peak of the sedimentgraph.

The term "dilution factor" in this study is defined, for practical and convenient purposes, as the ratio of suspended-sediment concentration on the construction site to that of a specific off-site downstream station at the peak time of the runoff hydrograph at that location. The specific downstream station refers to a point in the immediate vicinity where the drainage system enters the natural drainage watercourse. This requirement is necessary to assure that the construction site of interest is similar to the present case, with a relatively short-time concentration for surface runoff. The dilution factor is a lump-sum representation of the of the characteristics of the sediment-transport pattern in a drainage area

FIGURE 6

Data for Streamflow and Suspended Sediment Concentration due to Light Rain, January 4, 1975

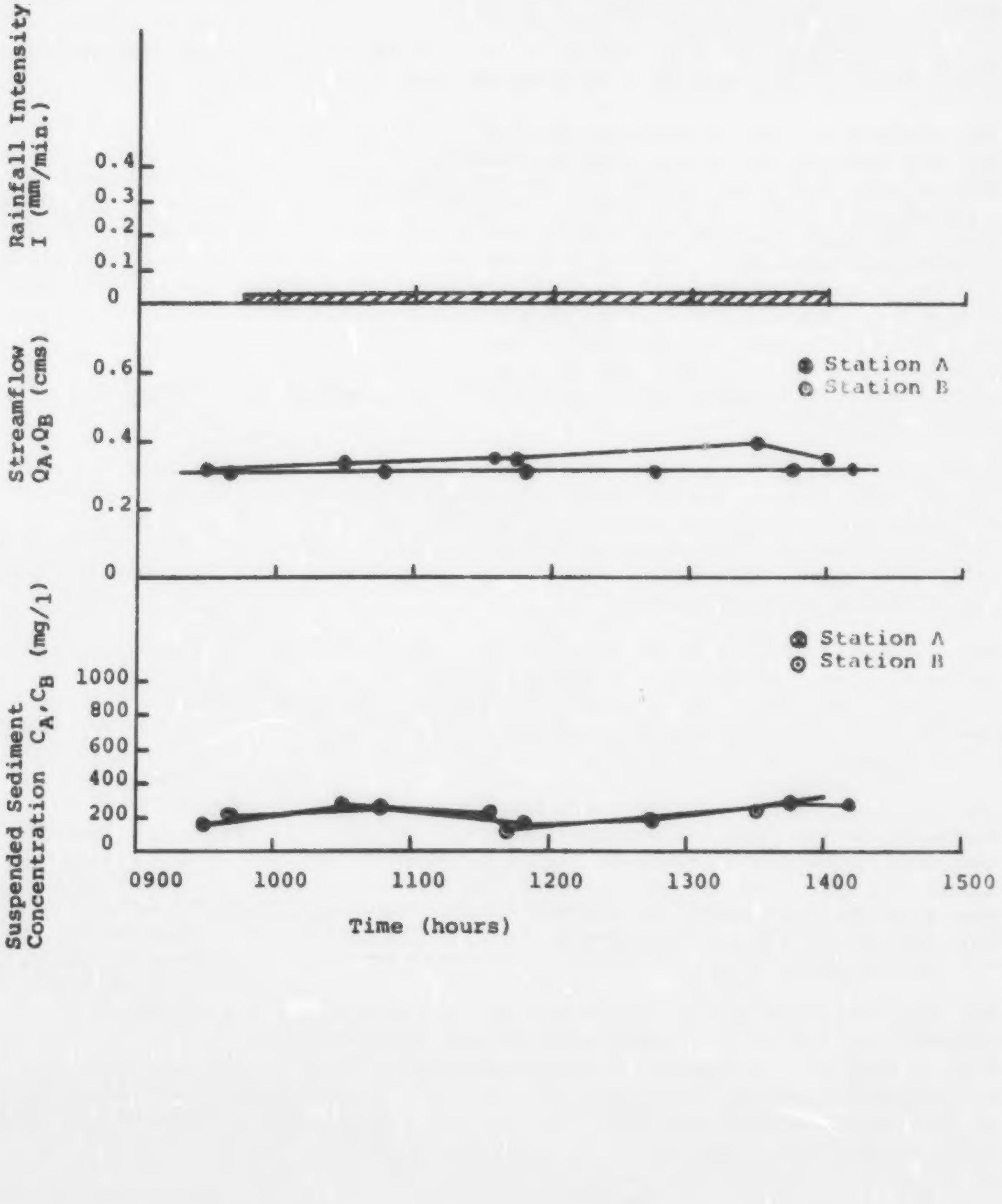


FIGURE 7

Time Variation of Suspended Sediment Concentration at Street Drain Inlets

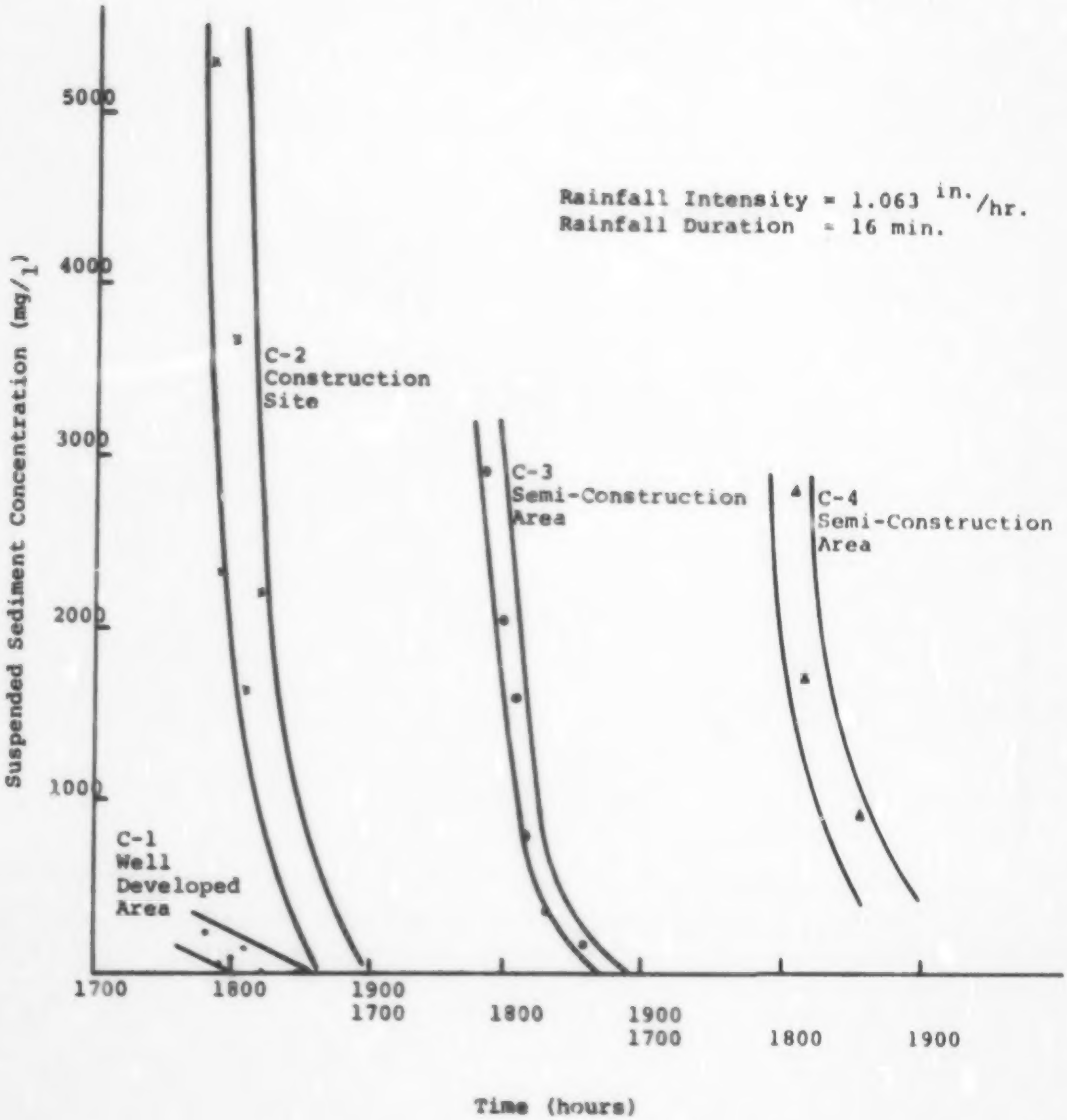
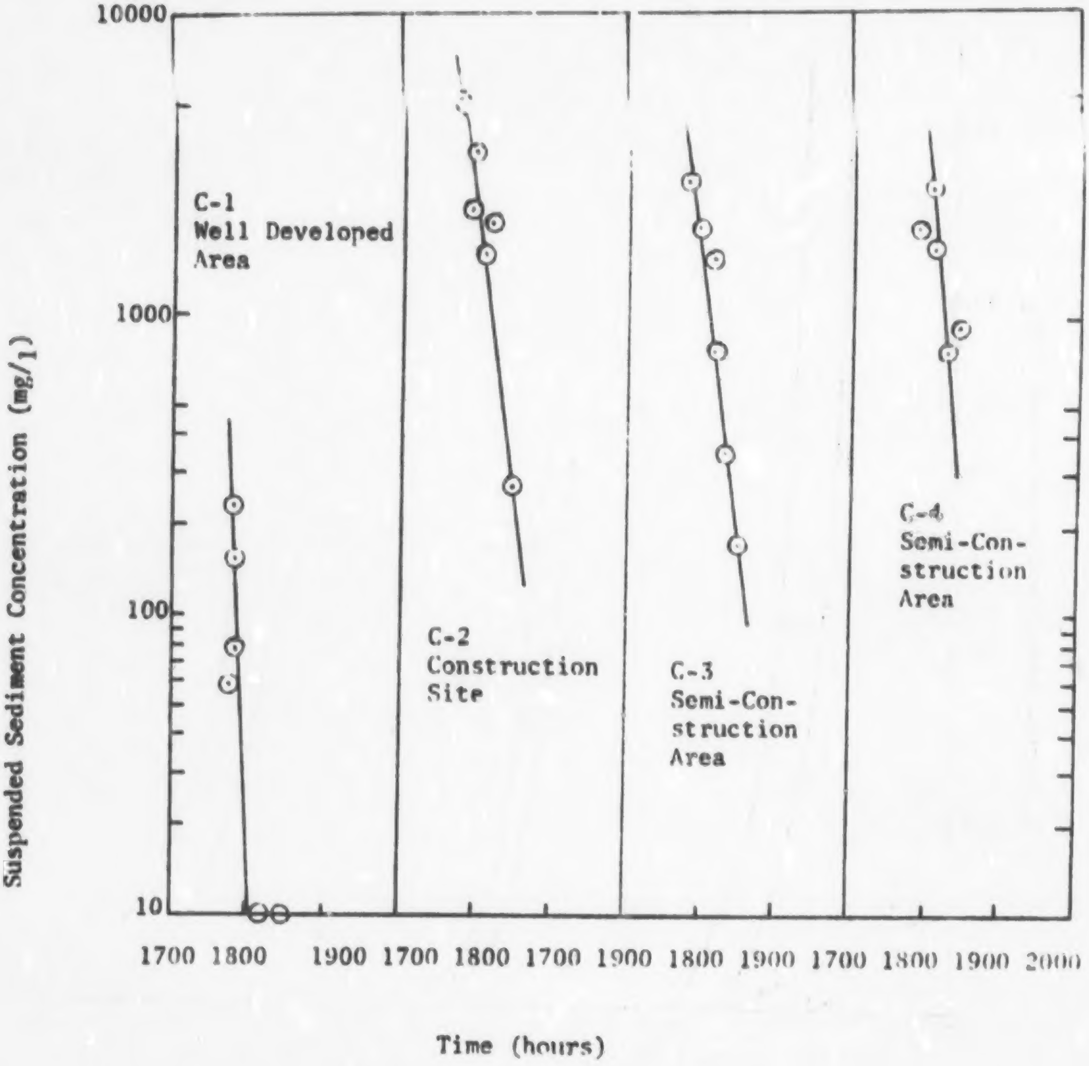


FIGURE 8

Time Rate of Change of Suspended Sediment Concentration at Street Drain Inlets



undergoing construction. It is evident that the dilution factor may be a function of many parameters related to the sediment delivery processes, such as drainage system, sediment properties, rainfall, and runoff. The correlation between the dilution factor and these parameters needs further investigation.

Figure 9 shows a simple relation between the dilution factor and rainfall intensity for nine storm events. The general tendency is for the dilution factor to increase as rainfall intensity decreases. For high-rainfall intensity, the dilution factors are expected to approach unity. The scatter in Figure 9 may be due to inexact timing of sampling, and possibly some difficulty with the definition of the dilution factor. Additional data is needed for analysis and possible derivation of a correlative functional equation.

In addition to the data just shown for the construction site, samples collected at a street-drainage inlet in each of the semi-construction and unconstructed sites are similarly, analyzed and the results presented in Figure 10 in a slightly different form with suspended-sediment concentration versus rainfall intensity. The concentration for each storm and the sample location varies as rainfall intensity changes. The average proportionality among the three concentrations for the three sites for a particular rainfall intensity is about 40:10:1. Concentrations at the three different sites for a given rainfall intensity are not the same, even though each site had the same erosive energy for that rainfall event. One factor producing this proportionality is the distinctive surface coverage at three sites. Based on this analysis, the proportionality for the soil erosion control practice factor, C, in the modified Universal Soil Loss Equation for three sites is thus estimated as $C_1:C_2:C_3 = 0.025 : 1.0 : 0.25$. Subscripts 1, 2, and 3 refer to nonconstruction, construction, and semi-construction sites, respectively.

The modified form of the Universal Soil Loss Equation as suggested by Chen [1974] is as follows:

$$q_c = RKLSC$$

where q_c = the rate of soil erosion from a construction site
R = the rainfall erosive energy
K = the soil erodibility
LS = the length of steepness of slope
C = the control practice

The C factor can be evaluated as the product of the control factors associated with each of the individual sediment-control measures. Thus:

$$C = C_s \cdot C_r \cdot C_t \cdot C_e \cdot C_o$$

where C_s = the control factor due to surface stabilizing or protecting treatment
C_r = the control factor due to runoff reduction practices
C_t = the control factor due to sediment trapping measures
C_e = the control factor due to restricting the spatial and/or temporal exposure of the denuded site to the rainfall and runoff erosion
C_o = other practices that are not included

The Virginia Beach Erosion and Sediment Control Ordinance [1973] suggests a value

FIGURE 9

Characteristics of Dilution Factor

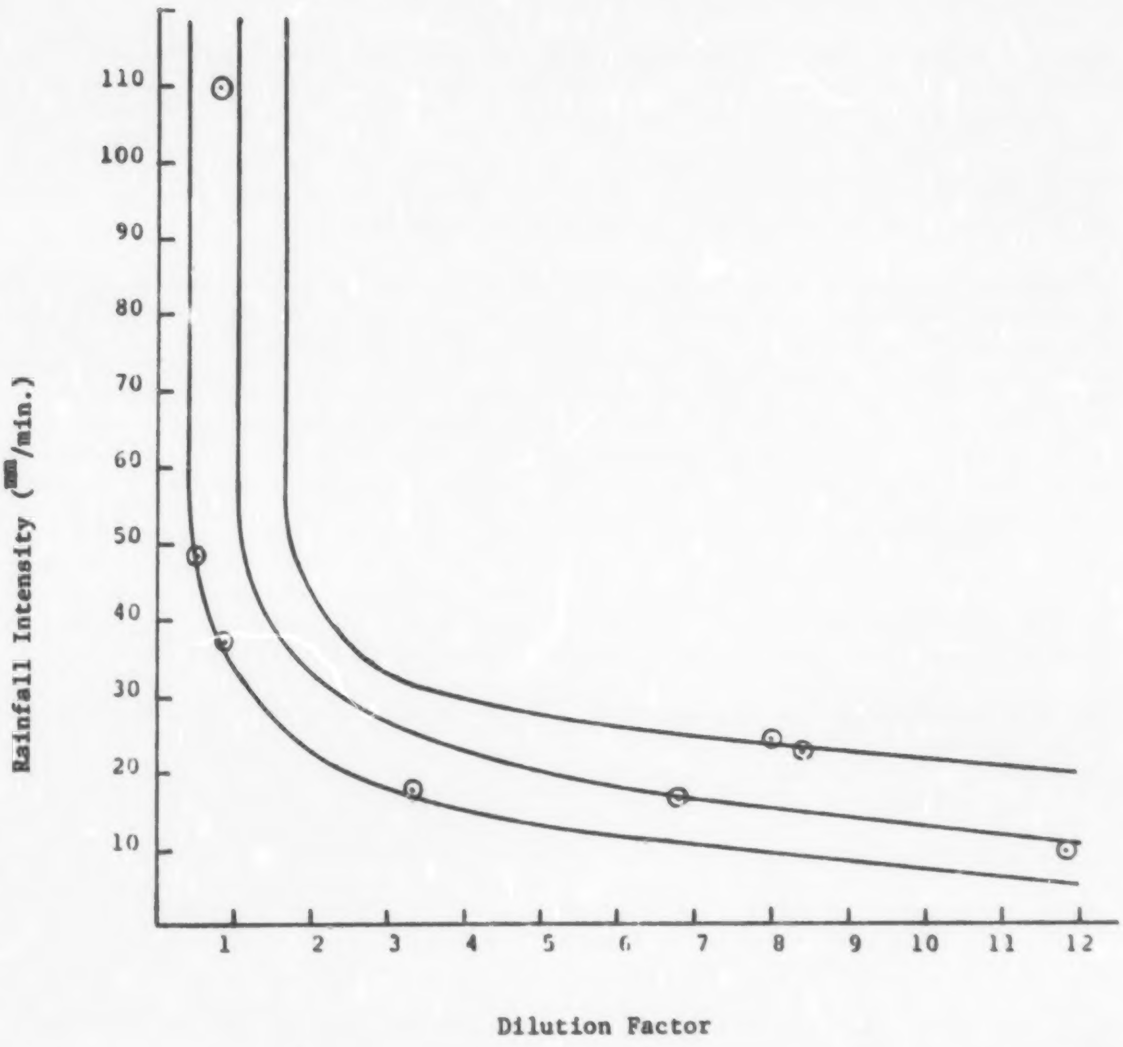


FIGURE 10

Spatial Variation of Suspended Sediment Concentration at Street Drain Inlets

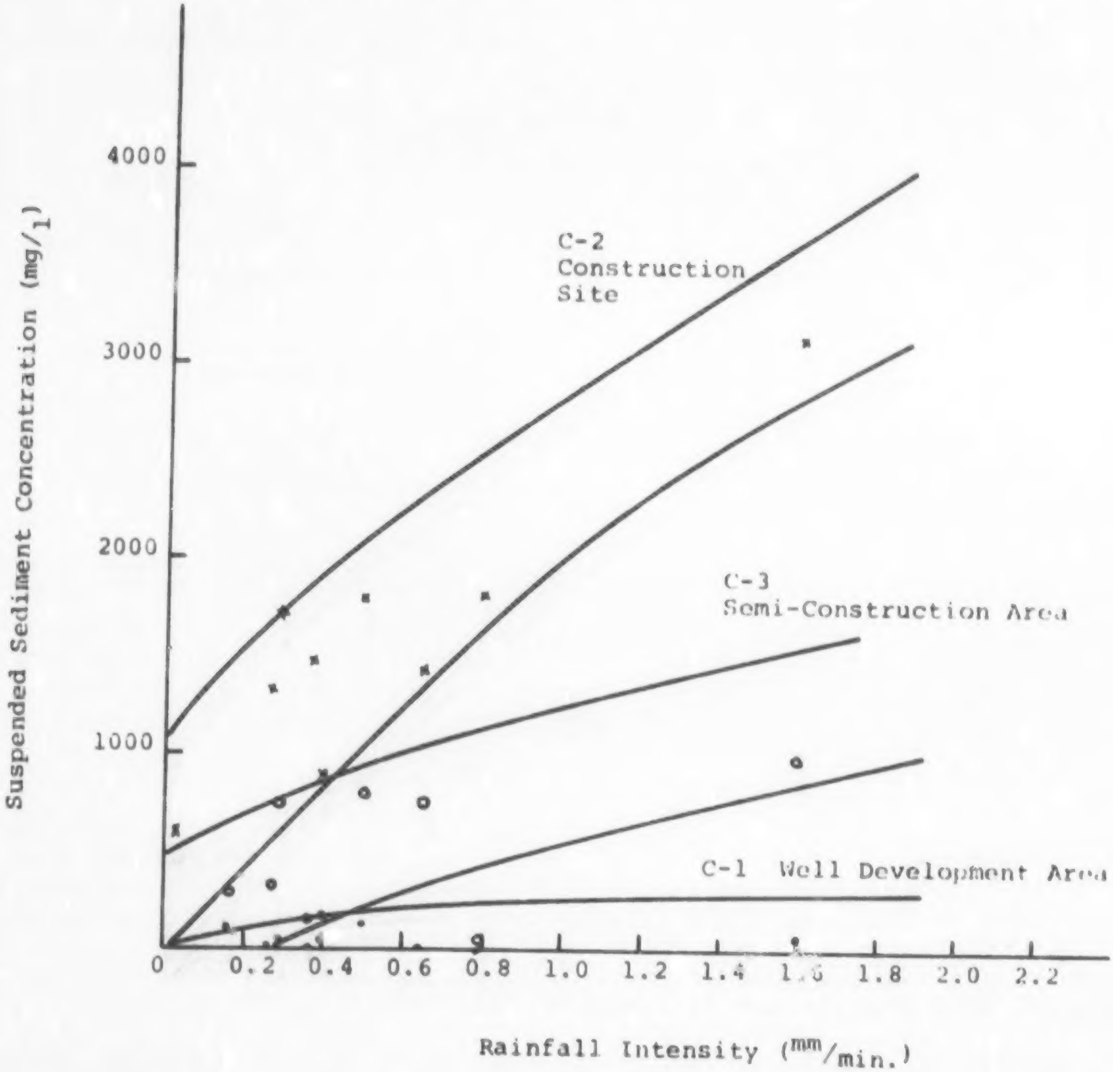
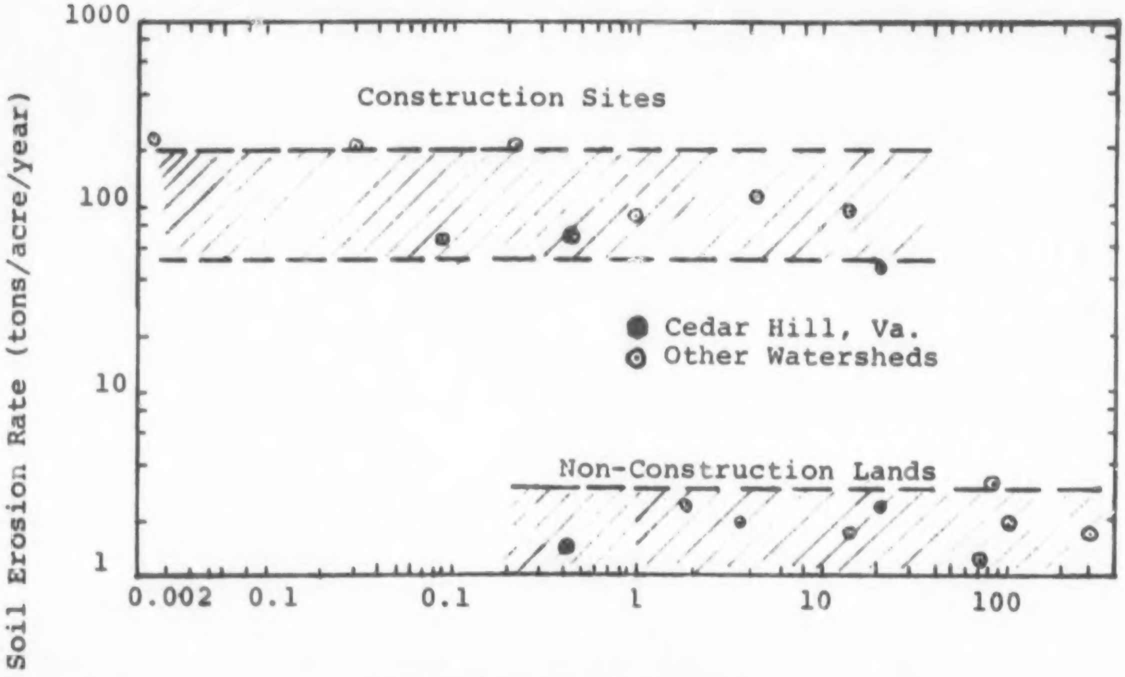


FIGURE 11

Soil Erosion Rates from Construction and Non-Construction Lands



Drainage Area (sq.mi.)

for average annual erosive energy of the rainfall (R) to be R = 300. The K value for the C horizon of Othello soil is 0.28. Because no sediment-control measures were taken in this case, the C value is taken as unity at the construction site. Using slope length of 50 feet and slope of 10 percent for each lot, the erosion rate is 65.7 tons per acre per year. This soil erosion rate, illustrated in Figure 11, falls into the same data band as indicated by Chen [1974].

To estimate the lump-sum control practice factor, C, for the semi-exposed area previously mentioned, the area is broken into two categories. One is for lawns and gardens (40.28 percent of the drainage area) and the other is the stripped but unconstructed land with some degree of vegetation growth (24.14 percent of the area). It is assumed that soil erosion from impervious surfaces, ponds, and woods is practically none. The C values for lawns and unconstructed areas can be deduced from the previous results as 0.25 and 0.025 respectively. It follows that soil-erosion rates from these two types of surface coverage are 16.4 tons per acre per year for semi-constructed area, and 1.6 tons per acre per year for well-developed area. The erosion rate for the well-developed area falls into the data band of non-construction lands shown in Figure 11.

The sediment yield, Q_s, in tons, at a downstream location (Station B) can be expressed as:

$$Q_s = D\left[q_c\left(\frac{A_c}{A}\right) + q_s\left(\frac{A_s}{A}\right) + q_u\left(\frac{A_u}{A}\right) + q_o\left(1 - \frac{A_c + A_s + A_u}{A}\right)\right]A$$

where D = the sediment delivery ratio defined as the ratio of the percentage of sediment delivered to a specific location in watershed to the soil erosion from the source area.

q_c, q_s, q_u, q_o = the soil erosion rates in tons per acre per year, from construction site, semi-construction site, nonconstruc-site, and other surfaces ($q_o = 0$).

A_c, A_s, A_u, A = the area in acres of construction site, semi-construction site, nonconstruction site, and total drainage area.

Based on the recommended curves for the sediment-delivery ratio as a function of drainage area for Maryland and the southeastern U.S. [Chen, 1974], the sediment yield is estimated as 10,300 tons per square mile per year when the sediment-delivery ratio is estimated to be 45 percent. This indicates that the drainage area undergoing construction is in the high-dilution category as classified by Wolman and Schick [1967] and illustrated in Figure 12.

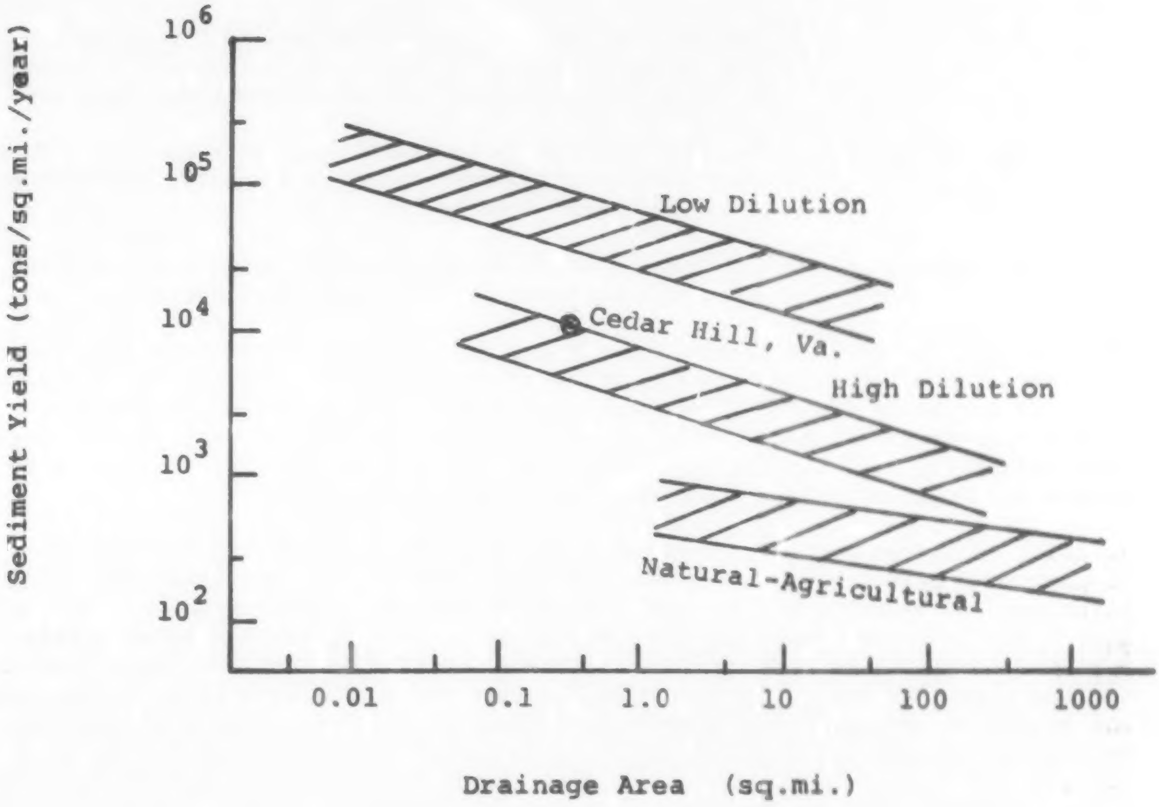
In order to better understand the sediment delivery ratio for this area, the thunderstorm of March 30, 1975, was analyzed. Rainfall intensities were 0.132 in/hr for the first 60 minutes and 1.84 in/hr for the next 30 minutes, this being equivalent to a recurrence interval of one year. The rainfall erosion energy for this single storm event was calculated to be 11.9, and soil-erosion rate at the construction site was estimated to be 2.6 tons per acre. Data on water discharge and suspended-sediment concentrations observed at Station B indicated that the sediment yield was 6.0 tons, and that the sediment-delivery ratio was only 3.2 percent. This discrepancy between the estimated and observed value of the ratio might be due to the occurrence of five rains of moderate to heavy intensity during the previous two weeks and the fact that the construction site was not newly disturbed during the period. This implies that the soil-erosion rate sometimes may be overestimated with the use of the Universal Soil Loss Equation. It seems evident that the impact of the previous rainfall events on soil conditions, such as compactness and moisture content, must be taken into account in computations to estimate the soil-erosion rate. A study of sediment-delivery ratios for other storms based on computed rainfall erosion energy indicated that the ratios ranged from 0.14 percent to 9.5 percent. The sediment-delivery ratios, as computed by the modified Universal Soil Loss Equation, are relatively low in the study area compared with other watersheds described in the literature [Chen, 1974].

Spring and fall seem to be the best seasons for construction activities in the study area. This will be verified by more data from continuing monitoring throughout the study. Indications of the seasonal variation of soil erosion rates should be helpful in determining best construction scheduling.

In this study area, periodic cleaning of the sediment remaining along the gutters

FIGURE 12

Effect of Construction Intensity and Drainage Area on Sediment Yield (From Wolman and Schick, 1967)



appears to be an easy control practice, but putting the sediment right back along the curbs is not recommended.

SUMMARY

The soil erosion rate at Cedar Hill, Virginia Beach, is estimated to be 65.7 tons per acre per year. The sediment yield is estimated as 10,300 tons per square mile per year. The observed sediment yield for a single storm is, in general, less than the estimated value, with the sediment delivery ratios ranging from 0.14 percent to 9.5 percent. The proportionality for soil-erosion rate among construction, semi-construction, and well-developed sites is found to be 1 : 0.25 : 0.025 based on an analysis of sediment samples collected at street-drainage inlets. The dilution factor was found to increase as rainfall intensity decreases. The dilution factor and sediment-delivery ration parameters in the coastal plain need further investigation in order to understand the impact on streams of sediment yield from construction sites.

ACKNOWLEDGEMENT

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CHARACTERIZATION OF
URBAN LAND RUNOFF

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INTRODUCTION

In urban areas, untreated point sources pose the greatest single threat to water quality. Consequently, point sources have long been studied, and much has been learned which can and is being used to diminish or in some cases eliminate the influence of point waste sources on water quality. Indeed, improved plant performance has become virtually synonymous with increased water quality. Unfortunately, better treatment plants do not always produce proportionate improvement in overall water quality. A sewage treatment plant, however sophisticated, can only treat that portion of the total urban pollution load it receives. While point waste sources are treated and are becoming less threatening, non-point sources of water impairment become relatively more significant.

In pursuit of national water-quality goals, various governmental units have raised standards for municipal and industrial waste treatment. The expenditure of public and private funds to obtain small-efficiency improvements increases exponentially as the quality of the effluent approaches purity. It appears, however, that current national goals for water quality may not be reached in some areas, even with pure municipal and industrial effluents. Whipple [1973] states that 90 percent of the pollution in a number of New Jersey watersheds did not originate from known effluent discharge locations. A recent report of the Council on Environmental Quality indicated that, in 80 percent of the urban areas studied, downstream water quality was not controlled by point sources [Bowen, 1972]. The Council also has determined that unrecorded pollution sources are so important that even if all recorded discharges of municipal and industrial wastes were eliminated, certain waters would still be polluted [Council on Environmental Quality, 1973].

In 1971, the U.S. Environmental Protection Agency and the North Carolina Water Resources Research Institute jointly funded a two-year research project at North Carolina State University for the purposes of:

1. Characterizing urban land runoff with respect to pollutant yield, and

2. Developing criteria for ascertaining the relative impact of point and non-point urban waste sources on water quality management.

This paper is a summary of portions of that project [Colston, 1974].

TABLE 1

Hydrologic Description of Urban Runoff Events Sampled

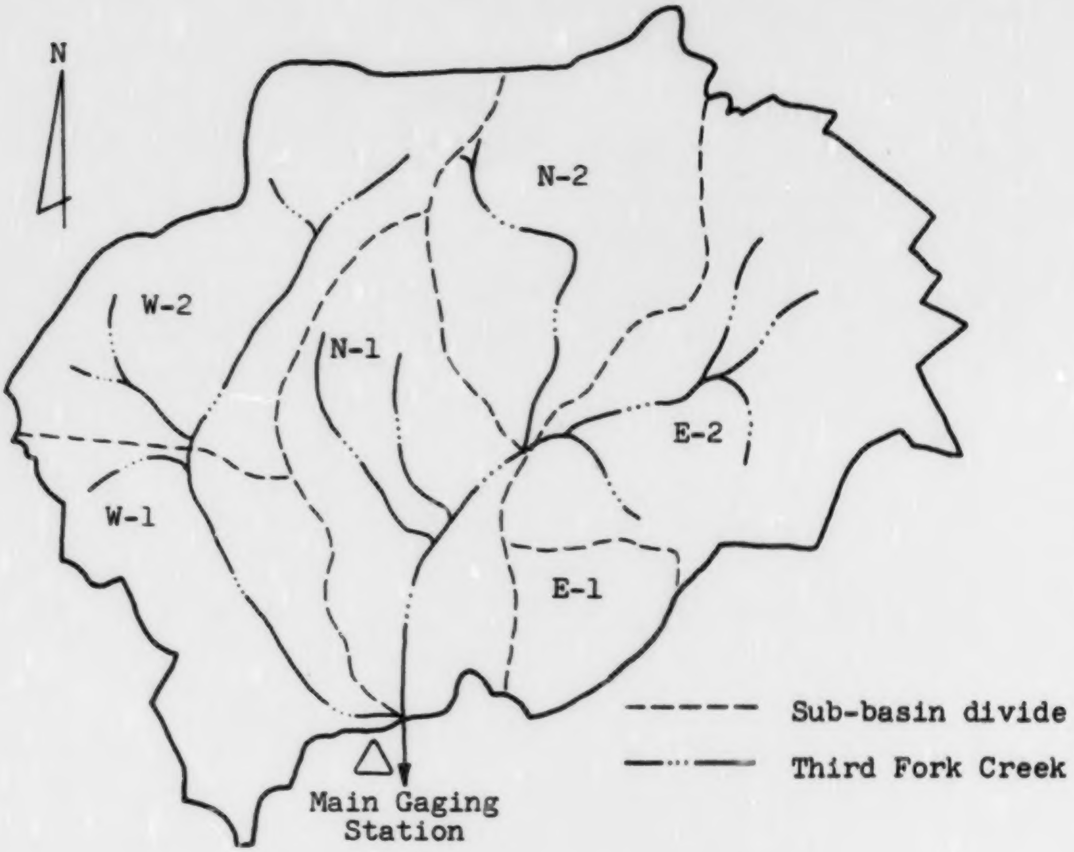
Date	Storm No.	Rainfall Inches	Duration Hours	Intensity In/hr	Runoff Inches	Runoff Coefficient	Peak Discharge CFS	Days Since Last Storm	No. Samples Taken
10/23/71	1	1.55	32.5	0.047	0.88	0.54	33.2	3.25	15
11/24/71	2	← NO PRECIPITATION RECORDS AVAILABLE →						34.0	13
12/16/71	3	0.05	0.5	0.1	0.003	0.0061	2.5	4.0	10
12/20/71	4	0.43	19.5	0.022	0.15	0.34	31.3	0.5	16
1/4/72	5	0.2	2.5	0.08	0.04	0.19	22.6	4.75	9
1/10/72	6	0.55	12.0	0.046	0.19	0.34	63.0	1.0	19
2/1-2/72	7	1.19	10	0.119	0.84	0.7	138.4	11.5	27
2/12-13/72	8	0.96	10	0.096	0.54	0.56	126.6	9.0	20
2/18/72	9	0.44	8	0.049	0.2	0.45	32.0	5.5	27
2/23/72	10	0.13	0.5	0.26	0.04	0.29	22.0	5.5	8
2/26/72	11	0.19	0.5	0.38	0.03	0.18	19.0	2.83	23
3/8/72	12	0.04	0.083	0.48	0.01	0.25	4.3	4.88	15
3/16/72	13	0.6	10.33	0.058	0.36	0.59	51.8	7.25	23
3/31/72	14	0.46	11.33	0.04	0.15	0.33	40.6	9.5	23
4/12/72	15	0.33	2.17	0.15	0.12	0.35	73.0	4.25	17
5/3/72	16	1.14	7.25	0.15	0.47	0.41	135.7	21.0	21
5/14/72	17	0.71	8.0	0.089	0.29	0.41	109.0	5.5	24
5/22/72	18	0.92	15.5	0.059	0.513	0.56	349.0	5.0	9

TABLE 1 (continued)

Date	Storm No.	Rainfall Inches	Duration Hours	Intensity In/hr	Runoff Inches	Runoff Coefficient	Peak Discharge CFS	Days Since Last Storm	No. Samples Taken
5/30-31/72	19	0.25	10.0	0.025	0.03	0.12	29.9	5.62	8
6/20/72	20	0.24	6.5	0.037	0.07	0.29	75.4	20.5	16
6/28/72	21	1.78	2.13	0.83	1.55	0.87	1740	7.17	5
7/11/72	22	0.1	0.5	0.2	0.005	0.054	2.25	6.54	4
7/12/72	23	0.33	3.83	0.086	0.083	0.25	36.2	7.33	15
7/17/72	24	0.26	1.0	0.26	0.15	0.57	125.0	5.25	7
7/31/72	25	0.38	2.5	0.15	0.34	0.9	152.0	0.75	20
8/28/72	26	0.06	2.1	0.028	0.004	0.006	2.58	6.54	3
9/17/72	27	1.51	3.3	0.45	0.7	0.46	700.0	11.3	10
9/21/72	28	0.5	9.0	0.055	0.083	0.16	41.4	3.5	10
10/5/72	29	2.36	26.0	0.34	2.07	0.88	872.0	5.0	7
10/19/72	30			← RECORDERS INOPERABLE →					11
11/14/72	31	0.74	3.63	0.2	0.25	0.34	120.8	6.0	9
11/19/72	32	0.79	4.0	0.19	0.48	0.61	106.0	2.45	20
11/30/72	33	0.5	14	0.04	0.09	0.18	57	4.6	12
1/19/73	34	0.11	1.25	0.09	0.03	0.30	27.8	4.2	26
2/26/73	35	0.4	1.6	0.25	0.09	0.24	83	12.1	3
3/21/73	36	0.25	5.0	0.05	0.05	0.23	38	4.1	16

FIGURE 1

Third Fork Creek Drainage Basin, Durham, North Carolina



STUDY AREA

The upper Third Fork Creek drainage basin selected for study is located in south central Durham, North Carolina. Durham is on a divide, the northern portion draining into the Neuse River system, the southern into the greater Cape Fear system. The upper Third Fork Creek has a 1.67 square-mile (1,093 acres) drainage basin completely within Durham city limits, with its northern boundary located in the downtown business section. The basin is primarily composed of two shallow valleys with relatively narrow flood plains located along the lower portion of each. Excess surface waters in the basin flow into the headwaters of Third Fork Creek primarily through natural channels. The stormwater runoff system is composed of overland flow, street gutters, small pipes, and culverts under roads. No proper storm sewer system exists; therefore, excess surface waters generally follow natural drainage patterns, except for a small part of the northern edge of the drainage basin located in the downtown business district.

A great diversity in land use, apparent personal income, and physical basin features occur within the urban Third Fork Creek basin [Figure 1]. Sub-basins W-1, W-2, E-1, and N-1 are primarily residential areas. The affluent residential section is W-1, where homes are in the $40,000 to $150,000 range. The least affluent sections are in sub-basins E-1, E-2, and N-2, where slums exist. Most streets in E-1 and E-2 are unpaved. Portions of N-2 and E-2 are undergoing urban renewal. Sub-basin N-2 contains downtown business property including light to heavy industry and a cross-town expressway. Sub-basins E-1, E-2, and W-1 contain essentially no industry, business, or commercial property. A small shopping center, a large park surrounding the flood plain, and a few middle-income homes typify N-1. Sub-basin W-2 contains business and commercial property, moderate income housing, and a large cemetery. Population density within the basin varies from 1.5 per acre in N-2 to 13.5 in E-1 with an average of 6.0 per acre. Land use in the basin is as follows: 59 percent residential, 19 percent commercial and industrial, 12 percent public and institutional with 10 percent unused. Impervious areas constitute 32 percent of the total land area.

SAMPLING

The United States Geological Survey operates a continuous stage recorder and two digital punch tape recorders within the 1.67 square-mile drainage basin. Samples were obtained automatically during runoff events by the use of a modified vacuum sampler, a submersible pump, a plexiglass sampling flume, and a control box. The control box turned the sample train on upon sensing a rise in the water level and obtained discrete samples at predetermined time intervals. The submersible pump was located adjacent to the stream bed procuring samples from the bottom flow regime during runoff events. Although the system was subject to clogging, it was judged to be reasonably reliable under the adverse conditions of sampling during high flows.

Thirty-six separate runoff events were sampled during the project period producing a total of 501 separate samples. A summary of information describing the hydrologic and time parameters associated with each storm sampled is presented in Table 1. The average, standard deviations and range of all urban runoff samples are presented in Table 2.

BOD DIFFICULTIES

The Biochemical Oxygen Demand (BOD) test is, and has been, the prime tool of engineers and chemists in estimating the amount of potentially biodegradable material present in a waste and the rate at which oxygen will be utilized in a receiving watercourse. During this project it became apparent that BOD was an inappropriate analytical test for organic characterization of urban land runoff. The major difficulty encountered with the BOD test was that results were affected by the percent dilution of the sample. The more dilute the sample, the greater the BOD exerted. Figure 2 presents typical variations of 5-day BOD as a function of dilution. The exact cause for the difficulties

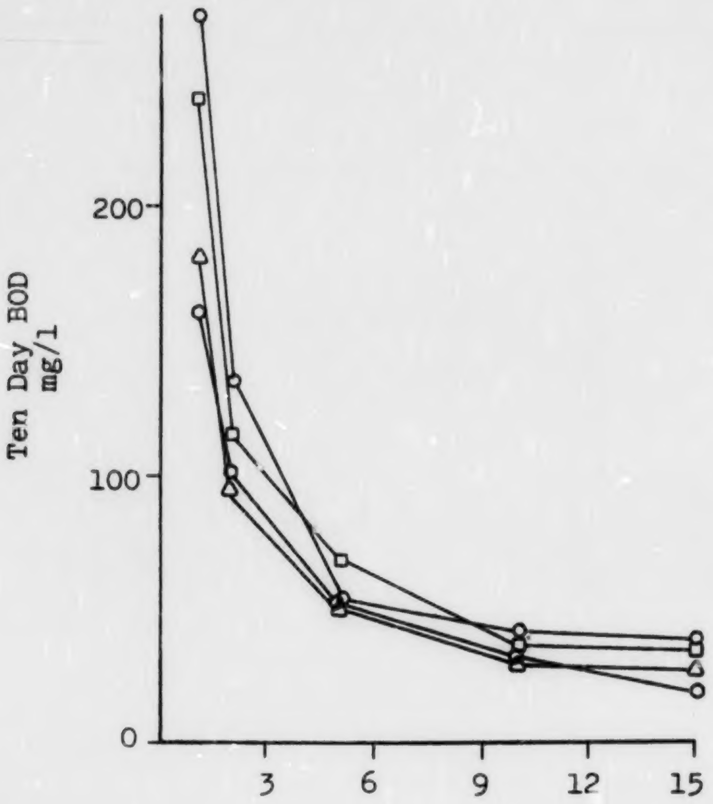
TABLE 2

Average, Range, and Standard Deviation of Pollutant Concentrations for All Storm Samples

Pollutant	Mean mg/l	Standard Deviation	Range (mg/l) Low	High
COD	170	135	20	1042
TOC	42	35	5.5	384
Total Solids	1440	1270	194	8620
Volatile Solids	205	124	33	1170
Total Suspended Solids	1223	1213	27	7340
Volatile Suspended Solids	122	100	5	970
Kjeldahl Nitrogen as "N"	0.96	1.8	0.1	11.6
Total Phosphorus as "P"	0.82	1.0	0.2	16
Fecal Coliform (#/ml)	230	240	1	2000
Aluminum	16	8.15	6	35.7
Calcium	4.8	5.6	1.1	31
Cobalt	0.16	0.11	0.04	0.47
Chromium	0.23	0.10	0.06	0.47
Copper	0.15	0.09	0.04	0.50
Iron	12	9.1	1.3	58.7
Lead	0.46	0.38	0.1	2.86
Magnesium	10	4.0	3.6	24
Manganese	0.67	0.42	0.12	3.2
Nickel	0.15	0.05	0.09	0.29
Zinc	0.36	0.37	0.09	4.6
Alkalinity as $CaCO_3$	56	30	24	124

FIGURE 2

Typical BOD Variations of Urban Land Runoff as a Function of Dilution



experienced with the BOD test as applied to urban land runoff is not known. However, they are believed to be due to (1) the inhibitory effect of heavy metals; (2) the presence of other unidentified inhibitory compounds, and/or (3) inherent problems of the standard BOD test.

COD EXERTION RATE STUDIES

One important aspect of this project was to determine the impact of urban runoff waste on a receiving stream by its effect on dissolved oxygen concentrations. In an attempt to provide needed information on the total amount of biodegradable material in urban runoff and to evaluate the exertion rate of the oxygen demand, the following study was initiated. Four liters of 100 percent urban runoff were placed in an Erlenmeyer flask. A stirring magnet was added and the flask placed in a dark 20°C incubation room. The four nutrients normally added

TABLE 3

Oxygen Uptake Rates (K_1) for Urban Land Runoff

Sample	Test	Percent Dilution	Rate** K_1	Ultimate Uptake mg/l	Percent* Biodegradable
Urban Runoff	COD	100	0.08	106	44
	BOD	0.5	0.07	540	
	BOD	1.0	0.13	190	
Secondary	COD	100	0.14	48	53
Sewage	BOD	1	0.20	125	
	BOD	5	0.10	55	
Urban Runoff	COD	100	0.20	98	53
	BOD	1	0.27	137	
	BOD	5	0.27	112	
	BOD	5	0.24	124	
Urban Runoff	COD	100	0.06	20	16
	BOD	2	0.16	47	
	BOD	5	0.20	30	
Urban Runoff	COD	100	0.13	57	61

* Biodegradable COD * Total Initial COD
** Base-e

to BOD dilution water were added in the same proportion to the four liter-samples. The stirring kept dissolved oxygen concentrations near saturation and the sample homogeneous. COD's (COD_I) were run on each sample initially and then periodically during following days at time "t" (COD_T). The difference between the initial COD_I and the COD of the sample at a later time (COD_T) can be attributed to biological activity or biodegration of the waste. The total amount of organic material degraded should represent the "ultimate BOD" as supposedly given by the standard BOD test. The COD jar test was continued for 20 days. The method of moments was employed for evaluation of K_1, a COD exertion-rate constant. The percent of the total COD susceptible to biodegradation was computed also. The oxygen uptake rate, K_1, for urban runoff as determined by the method of moments from BOD and COD uptake data is presented in Table 3.

A sample of secondary effluent from a municipal waste plant prior to chlorination also is included in the table for comparative purposes. The K_1 rates as determined by both techniques compare favorably, indicating that either could be used. The ultimate oxygen demand in 20 days as determined from the COD tests of 100 percent sample was always less than that predicted by the BOD test. It appears that the K_1 rate is fairly independent of sample dilution or analysis technique, whereas the 20-day ultimate oxygen demand is very dependent on sample dilution if determined by the BOD techniques. The percent of COD capable of biodegradation in 20 days was found to vary from 16 to 61 percent, with an average of 44 percent. From these studies it is concluded that: (1) approximately 40 to 50 percent of the COD load contributed by urban runoff is susceptible to biodegradation in 20 days; (2) the COD K_1 exertion rate is equivalent to that obtained by a conventional BOD test; (3) the standard BOD test is not an appropriate test for evaluation of urban land runoff, and (4) the COD uptake test gives reproducible values at 100 percent dilution.

REPRESENTATIVE SAMPLING

The automatic sampling device located at the USGS gauging station was powered by a submersible centrifugal pump on the bottom of the stream channel as previously described. Consequently, during runoff events the samples taken were from the lower portion of flow in the stream. The question arose as to the representativeness of the samples procured in relation to the average "true" concentration of pollutants at any given time. In an attempt to define the magnitude of the pollutant variations within the cross-sectional flow, samples were obtained manually at the surface of the stream on three separate storms at the same time as the automatic sampler was procuring its sample. The average pollutant concentration at just below the water's surface is compared with the average pollutant concentration of the corresponding samples obtained by the automatic sampler in Table 4. The concentration of pollutants was almost always greater for samples obtained automatically (adjacent to the bottom) than they were at the surface.

POLLUTANT REGRESSION EQUATIONS

In order to describe pollutant concentration variations within storms and to assess the annual pollutant yield of urban runoff, analytical data from the 36 storms sampled were used to determine appropriate regression equations relating yield to runoff characteristics. The independent variables used were rate of runoff (CFS), time from storm start (TFSS) in hours, time from last storm (TFLS) in hours, and time from last peak (TFLP) in hours.

Initially, all four independent variables were used for regression to determine which were most significant in describing pollutant yield variations within storm events. From this, it was found that the rate of discharge (CFS) and time from storm start (TFSS) in hours were the two most significant variables. Only a modest gain in the correlation coefficient, r^2, was realized within the inclusion of the additional two time variables. It was, therefore, decided to limit the regression equations to CFS and TFSS for regression simplicity.

Prior bias would indicate that TFLS should be a significant factor, since the more frequently runoff events occur, the less buildup time there is for pollutants on an urban watershed. However, for the Third Fork Creek in Durham, the frequency of runoff events did not appear to influence significantly the pollutant discharge from the basin. The final regression equations describing the urban runoff pollutant flux in pounds per minute as a function of CFS and TFSS are presented in Table 5 with the corresponding correlation coefficients.

TABLE 4

Comparison of Pollutant Concentrations Adjacent to Waters Surface with Those Obtained by Automatic Sampler

Pollutant	Surface Concentration	Automatic Sampler Concentration	Surface Concentration As a % of Automatic Sampler Concentration
COD	150	233	67
TOC	40	53	75
BOD_5	62	62	100
K-Nitrogen	0.4	0.4	100
Total Phosphorus	0.78	0.90	87
Fecal Coliform (#/ml)	348	427	81
Total Solids	1413	2976	47
Volatile Solids	269	321	84
Total Sus. Solids	1156	2611	44
Volatile Sus. Solids	142	211	67
Calcium	1.76	1.69	104
Chromium	0.15	0.18	83
Iron	10.7	14.8	72
Magnesium	8.8	9.8	90
Manganese	0.81	1.11	73
Lead	0.24	0.31	77
Zinc	0.34	0.45	76

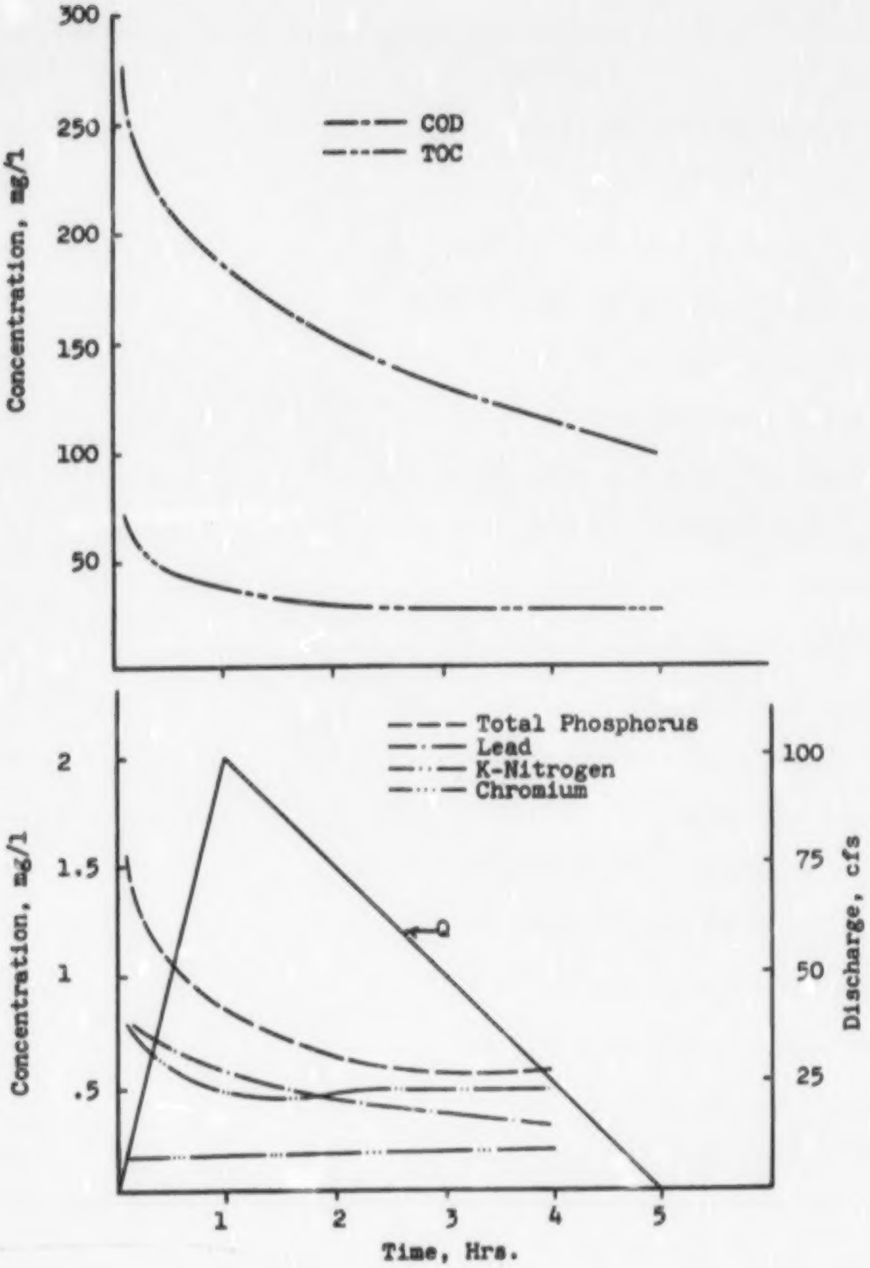
TABLE 5

Equations Describing Urban Runoff Pollutant Flux Near Channel Bottom in Pounds per Minute for Durham, North Carolina, as a Function of Discharge Rate (CFS) and Time from Storm Start (TFSS) in Hours

Equation	R_2
COD = 0.51 $\text{CFS}^{1.11}$ $\text{TFSS}^{-.28}$	0.90
TOC = 0.16 $\text{CFS}^{1.0}$ $\text{TFSS}^{-.28}$	0.84
Total Solids = 3.35 $\text{CFS}^{1.14}$ $\text{TFSS}^{-.18}$	0.85
Volatile Solids = 0.58 $\text{CFS}^{1.09}$ $\text{TFSS}^{-.11}$	0.92
Suspended Solids = 1.89 $\text{CFS}^{1.23}$ $\text{TFSS}^{-.16}$	0.76
Volatile Suspended Solids = 0.25 $\text{CFS}^{1.18}$ $\text{TFSS}^{-.17}$	0.83
Kjeldahl Nitrogen = 0.0032 $\text{CFS}^{.87}$ $\text{TFSS}^{-.29}$	0.73
Total Phosphorus as P= 0.003 $\text{CFS}^{1.03}$ $\text{TFSS}^{-.29}$	0.92
Aluminum = 0.0443 $\text{CFS}^{1.05}$ $\text{TFSS}^{-.15}$	0.89
Calcium = 0.045 $\text{CFS}^{0.60}$ $\text{TFSS}^{-.09}$	0.82
Cobalt = 0.0003 $\text{CFS}^{1.18}$ $\text{TFSS}^{+.13}$	0.92
Chromium = 0.0008 $\text{CFS}^{.96}$ $\text{TFSS}^{+0.06}$	0.89
Copper = 0.00035 $\text{CFS}^{1.10}$ $\text{TFSS}^{+.08}$	0.94
Iron = 0.0238 $\text{CFS}^{1.24}$ $\text{TFSS}^{-.18}$	0.87
Lead = 0.0013 $\text{CFS}^{1.125}$ $\text{TFSS}^{-.29}$	0.83
Magnesium = 0.0434 $\text{CFS}^{.98}$ $\text{TFSS}^{-.16}$	0.94
Manganese = 0.0023 $\text{CFS}^{1.11}$ $\text{TFSS}^{-.27}$	0.94
Nickel = 0.0005 $\text{CFS}^{1.03}$ $\text{TFSS}^{=.01}$	0.94
Zinc = 0.0011 $\text{CFS}^{1.10}$ $\text{TFSS}^{-.22}$	0.89

FIGURE 3

Pollutant Concentrations in mg/l for a Typical Storm Hydrograph as Predicted by Adjusted Regression Equations.



These equations may be adjusted to relate pollutant concentration in mg/l to rate of flow and time from storm start. The regression equations may also be adjusted to reflect concentrations at mid-depth. The equation expressing COD in pounds per minute as a function of CFS and TFSS is:

$$\mathrm{COD} = 0.51\ \mathrm{CFS}^{1.11}\mathrm{TFSS}^{-0.28}$$

where COD is in pounds per minute

CFS is cubic feet per second

and TFSS is time in hours from the initiation of runoff

This COD equation may be adjusted to reflect concentrations at average depth with the use of a correction factor derived from information contained in Table 4. The correction factor for COD is (1 + .67) ÷ 2 = 0.835. This yields the equation:

$$\mathrm{COD} = .425\ \mathrm{CFS}^{1.11}\ \mathrm{TFSS}^{-.28}$$

To modify the COD equation to reflect mg/l COD as a function of CFS and TFSS, the relationship is:

$$\mathrm{mg/l} = \frac{267\ \mathrm{lbs/min}}{\mathrm{CFS}}$$

The regression equation, therefore, becomes:

$$\mathrm{COD} = 113\ \mathrm{CFS}^{0.11}\mathrm{TFSS}^{-.28}$$

where COD is in mg/l at average depth.

All regression equations presented in Table 5 may be adjusted accordingly. Figures 3 through 5 present variations in pollutant concentrations as a function of CFS and TFSS as predicted by adjusted regression equations for a typical storm hydrograph.

ANNUAL POLLUTANT YIELD

Runoff data on each storm event occuring during the 1972 calendar year was obtained from discharge records at the Third Fork Creek gauging station. Thirty-minute-interval discharge rates for 66 storms was obtained. The time from the storm start, together with the rate of runoff and the regression equations, was used to determine the annual pollutant yield in pounds per acre per year as a result of urban runoff.

The annual pollutant yield of urban runoff was computed from base-flow periods and periods of urban runoff. The total pounds predicted by the regression equations based on data from the automatic sampler are adjusted to reflect the concentrations at mid-depth, assuming a linear vertical distribution. The annual yield of pollutants from urban runoff in pounds/acre/year for the urban

FIGURE 4

Pollutant Concentrations in mg/l for a Typical Storm Hydrograph as Predicted by Adjusted Regression Equations.

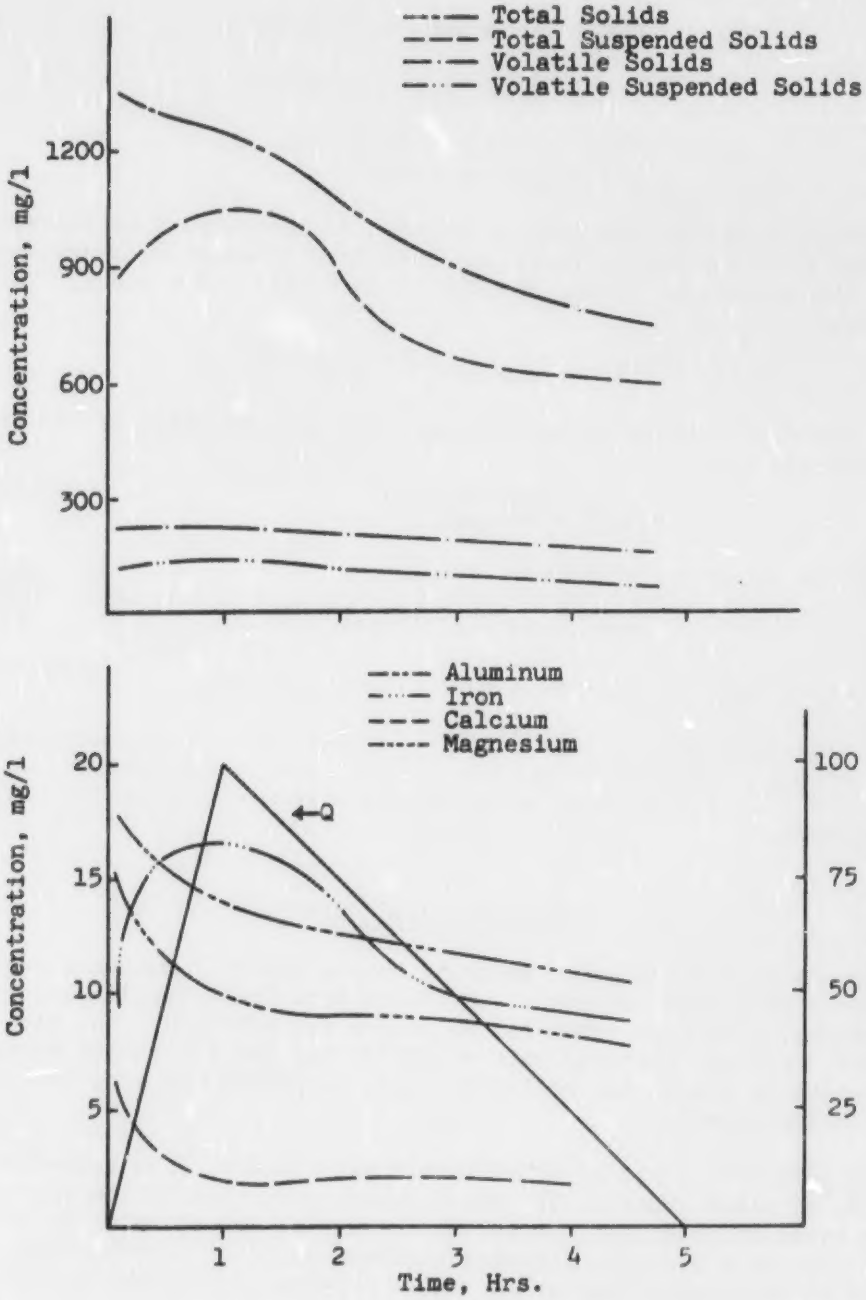


FIGURE 5

Pollutant Concentrations in mg/l for a Typical Storm Hydrograph as Predicted by Adjusted Regression Equations.

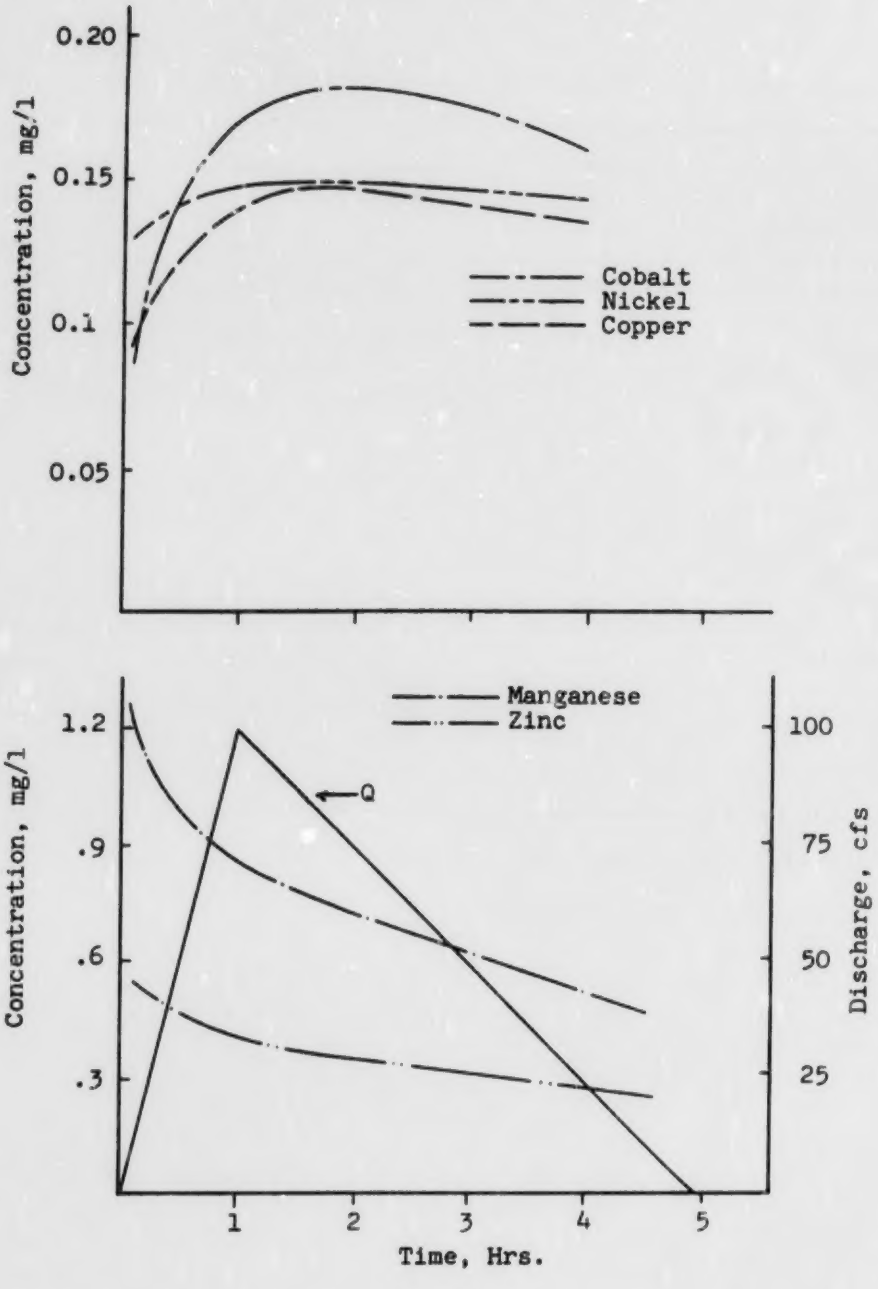


TABLE 6

Estimated 1972 Pollutant Yield for Third Fork Creek Drainage Basin in Pounds/Acre/Year

	Urban Runoff				
Pollutant	Predicted	Correction Factor	Adjusted Yield	Base Flow	Annual Yield*
COD	1071	0.835	895	43	938
TOC	190	0.875	166	21	187
Total Solids	9660	0.735	7100	600	7700
Volatile Solids	1440	0.92	1324	134	1458
Suspended Solids	9190	0.72	6617	74	6691
Volatile Sus. Solids.	910	0.835	760	37	797
Kjeldahl Nitrogen as "N"	2.8	1.0	2.8	3.3	6.1
Total Phosphorus as "P"	4.1	0.935	3.8	0.9	4.7
Aluminum	76	0.90**	63	1.5	64
Calcium	14	1.02	14	34	52
Cobalt	1.7	0.90**	1.5	0.4	1.9
Chromium	1.4	0.915	1.3	0.3	1.6
Copper	1.3	0.90**	1.2	0.4	1.6
Iron	118	0.86	100	2.2	102
Magnesium	57	0.95	54	17	71
Manganese	4.9	0.865	4.2	0.7	4.9
Nickel	1.1	0.90**	1.0	0.2	1.2
Lead	2.8	0.885	2.5	0.4	2.9
Zinc	2.1	0.88	1.8	0.2	2.0

* Annual Yield = Base Flow + Adjusted Yield

** Correction Factor Estimated

Third Fork study area is presented in Table 6. During the 1972 calendar year, base flow existed for 7,080 hours, or 81 percent of the time, while runoff from the 66 storms required 1,680 hours, or 19 percent of the time.

RELATIVE IMPACT OF URBAN LAND RUNOFF

The relative impact of urban land runoff on water quality is dependent on the physical and chemical characteristics associated with the particular aqueous system receiving the waste. As each receiving watercourse tends to differ, the impact--relative or absolute--is different. Consequently, each municipality must assess the magnitude of urban runoff for its particular situation. The annual pollutant yield of urban runoff was evaluated in comparison to municipal waste in terms of pounds and concentrations.

COMPARISON WITH DOMESTIC WASTE

The 1.67 square-mile study area is served by the Durham Third Fork Sewage Treatment Plant which received wastes from a total area of 9.6 square miles. The average daily waste volume during 1972 was 3.3 MGD, with average raw waste concentrations of 205 mg/l suspended solids, 285 mg/l 5-day BOD, 7 mg/l total phosphorus as "P," 4.4 mg/l nitrate nitrogen, 0.06 mg/l chromium, 0.12 mg/l copper, 0.9 mg/l zinc, 0.5 mg/l lead, and 0.1 mg/l nickel. The plant's average removal efficiency for BOD_5 and suspended solids was 91 and 85 percent, respectively. As a result of the long-term COD uptake rates described previously, it was determined that approximately 50 percent of the total initial COD of urban runoff could be biologically degraded in 20 days. Because of the difficulties experienced with running BOD tests on urban runoff, the ultimate BOD of urban runoff was assumed to be equal to the percent of the COD susceptible to biodegradation. As no COD tests were run at the Durham Sewage Treatment plant, it was assumed that the five-day BOD was 68 percent of the ultimate and that the COD of the raw municipal waste was 150 percent of the ultimate BOD. Therefore, the COD of the raw municipal waste was 2.2 (i.e., 1.5 ÷ 0.68) times the five-day BOD as measured.

Table 7 compares total quantities of raw municipal wastes and urban runoff, including base flow, in pounds per acre per year. The contribution of urban runoff reflects the adjusted contribution as described previously in Table 6. Urban runoff contains the majority of the heavy metals, varying from 57 percent of the total zinc yield to 94 percent of the chromium. It is important to note that if Durham provided 100 percent removal of organics and suspended solids from the raw municipal waste on an annual basis, the total reduction of pollutants discharged to Third Fork Creek would only be 52 percent of the COD, 59 percent of the ultimate BOD, and only 5 percent of the total suspended solids.

Table 8 gives the total annual yield of pollutants from municipal and urban runoff sources in pounds per acre during 1972, based on actual removal rates for the Durham Third Fork Sewage Treatment Plant. On a yearly basis, the average ultimate BOD reduction is 46 percent, COD reduction is 52 percent, and suspended solids reduction is 4 percent.

TABLE 7

Comparison of Raw Municipal Waste and Urban Runoff on an Annual Basis in Pounds per Acre per Year Pollutant Yield

Pollutant	Raw Municipal Waste		Urban Runoff + Base Flow		Total Annual Yield
	lbs	%	lbs	%	
COD	1027	52	938	48	1965
BOD Ultimate	685	59	470	41	1155
Suspended Solids	335	5	6690	95	7025
Kjeldahl Nitrogen as "N"			6.1		
Nitrate "N"	7.2				
Total Phosphorus as "P"	11	73	4.7	27	15.7
Chromium	0.10	6	1.6	94	1.7
Copper	0.20	11	1.6	89	1.8
Lead	<0.8	21	2.9	79	3.7
Nickel	<0.16	12	1.2	88	1.3
Zinc	1.5	43	2.0	57	3.5

Table 9 evaluates the total yield of pollutants from the Third Fork Creek watershed during only wet periods of urban runoff. The urban runoff contribution used to construct this table does not include pollutant yield during the 7,080 hours of base flow. During the 1,680 hours of wet weather, the raw municipal waste load represents only 18 percent of the total COD, 23 percent of the ultimate BOD, and 1 percent of the total suspended solids load. Consequently, if Durham provided 100 percent treatment of municipal wastes during these periods, it would represent an overall reduction of only 18, 23, and 1 percent of COD, ultimate BOD, and suspended solids, respectively. It should be noted that during approximately 20 percent of the time, downstream water quality is controlled by urban runoff.

TABLE 8

Total Annual Yield of Pollutants from Municipal and Urban Runoff in Pounds/Acre During 1972

Parameter	Municipal Waste Raw	Municipal Waste % Removal	Municipal Waste Effluent	Urban Runoff	Total Release	Overall Removal Efficiency
COD	1027	91*	92	938	1030	52%
Ultimate BOD	685	91	61	470	531	46%
Suspended Solids	335	85	50	6690	6740	4%

* Assumed

TABLE 9

Total Yield of Pollutants During Storm Periods from Urban Runoff and Raw Municipal Wastes in Lbs/Acre During 1972

Parameter	Raw Municipal Wastes	Urban Runoff	Total	Percent Municipal	Percent Runoff
COD	195	895	1090	18	82
Ultimate BOD	130	447	577	23	77
Suspended Solids	64	6617	6681	1	99

CONCLUSIONS

The conclusions of this report are based on information obtained in Durham, North Carolina, and are believed to be typical of urban areas of the Piedmont province on the East Coast. The specific conclusions of this two-year study are:

1. Urban runoff was found to be a significant pollutant source.

2. During approximately 20 percent of wet weather, downstream water quality was primarily governed by urban land runoff.

3. The concentration of organics in urban-level runoff are less than the concentration for typical raw wastes, whereas the concentrations of solids and heavy metals are greater in urban land runoff.

4. The standard Biochemical Oxygen Demand (BOD) test was not found to be an appropriate qualitative test for organics in urban-level runoff. This is believed due to the inhibitory effect of heavy metals and/or inherent problems with the standard test.

5. Approximately 40 to 50 percent of the total COD in urban land runoff is susceptible to biodegradation.

6. A vertical distribution of pollutant concentrations within natural urban drainage channels was found to exist, with pollutant concentrations typically increasing with depth from the surface.

7. In a regression analysis of urban runoff pollutant concentrations, the significant independent variables were found to be the rate of discharge (CFS) and the time from storm start (TFSS) as indicated by initiation of runoff.

8. A first flush of pollutants was found to exist in urban land runoff.

RECOMMENDATION

It appears that the impact of urban land runoff on water-quality management is significant and should be thoroughly evaluated before investments are made in upgrading existing secondary municipal waste plants.

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HEAVY METALS IN URBAN RUNOFF

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INTRODUCTION

During the past several years, there has been an increasing interest in the contamination of receiving waters with heavy metals. Concern over these materials stems primarily from their toxicity; as a result, elements such as mercury, lead, cadmium, chromium, and zinc have earned considerable notoriety.

Of particular significance to the consideration of heavy metals is the fact that they are non-degradable and hence persist in the environment for extended periods of time. In addition, heavy metals tend to precipitate out of solutions with relatively neutral pH values and some alkalinity, and may be adsorbed on clay particles or bound by such compounds as the hydrous oxides of iron and manganese. As a result, these materials are concentrated in the solid phases of water systems. Consequently, even though the water itself may contain only small amounts of these materials, the particulate matter in the water--and especially the benthic or bottom deposits--may contain considerable quantities.

Because metals are conservative, they may undergo biological magnification in the food chain, reaching concentrations in the upper trophic levels several orders of magnitude greater than those which originally existed in the water [Ayling, 1974]. Finally, depending on specific environmental conditions, certain metals such as mercury may undergo microbiological transformations to forms exhibiting significantly more toxicity [Gilles, et al., 1974; Dunlop, 1971].

Heavy metals enter receiving waters from a variety of sources. The rocks and soils directly exposed to surface water usually are the largest natural source. Dead and decomposing vegetation and animal matter also contribute small amounts of metals to the environment. Relatively large quantities of heavy metals such as lead and zinc have been observed in dry and wet atmospheric fallout [Bowman, et al., 1972; Daines, et al., 1970; Lazrus, et al., 1970; Morrow and Brief, 1971]. The majority of these materials have been associated with the combustion of fossil fuels and processing of metals [Rubin, 1974; Chow and Earl, 1970]. Other man-induced sources of heavy metals to the environment include domestic wastewaters, the industrial processes of plating, oil and coal refining, cleaning and medicinal agents, paints, catalysts, preservatives, and biocides.

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The purpose of this investigation was to evaluate more thoroughly the contributions of heavy metals resulting from urban runoff. Relatively little work has been devoted to this area as compared to the study of other pollutants [Bryan, 1971; Weibel, et al., 1964; Whipple, et al., 1974].

Newton, et al., [1974] presented observations of lead concentrations in street runoff. Utilizing data from automotive emissions and knowledge of daily vehicle travel in the Oklahoma City drainage basin, a theoretical value of .23 mg/l was calculated. This value was based upon the following assumptions: (1) fifty percent of the lead emitted by automobiles is deposited on the roadbed; (2) one-hundred percent of the lead on the roadbed is carried in the runoff, and (3) precipitation over the drainage basin is uniform.

Bryan [1974], in a study of the Durham, N.C. area, provided observational data that generally supported the Oklahoma City "theoretical value" relating to the amount of lead yielded by an urban drainage basin. The flow-weighted mean lead concentration for the 1,067 acre drainage basin studied was .059 mg/l. In terms of land use, it was observed that concentrations from the predominantly residential area were one-half of those from the more commercially developed areas, and were almost exactly in proportion to the percentage of paved surfaces.

Hem and Durum [1973] sampled runoff originating mostly from paved streets and other impervious surfaces in the city of Palo Alto, California. Lead concentrations of .093 mg/l were reported, .091 mg/l of which could be removed by filtration. Similar results were obtained in a study of the San Francisco Bay area.

Pitt and Amy [1973], in a study sponsored by the U.S. Environmental Protection Agency, attempted to determine the amount and nature of toxic materials of street surface contaminants, noting the effect of local factors on their distribution. Based on direct measure of street sweepings, the following average concentrations of storm-water runoff were calculated for a hypothetical city of 100,000 people during a one-hour rainfall of 0.1 inch of rain: Pb 6.2 mg/l; Zn 1.4 mg/l; Cu .37 mg/l; Cr .80 mg/l, and Ni .10 mg/l.

These data vary widely, although they certainly show that the amounts of heavy metals in urban runoff are quite significant. Concentrations of metals range from relatively low values to those equalling or surpassing concentrations found in secondary treatment-plant effluents [Table 2]. The difficulty with using much of this information in a quantitative way is that heavy metals concentrations in urban runoff are dependent upon a number of factors, some of which include:

1. The intensity and duration of the storm.
2. Length of antecedent dry period.
3. The general effectiveness of waste removal in the streets.
4. The composition of the storm sewer.
5. Land use.

As a result, sporadic or fragmentary data may give results widely different from those generally characteristic of the basin. Only data systematically gathered and classified with respect to population, industrial activity, and hydrologic background can be expected to be useful for planning purposes [Whipple, et al., 1974].

METHODS

Basin Characteristics

The site selected for our analysis of heavy metals in urban runoff was located in Lodi, New Jersey. Lodi is a relatively small community with a population of about 28,000. This area is urban and primarily residential in nature, with 62.4 percent of the area residential, 10.2 percent commercial, and 27.4 percent industrial. There are several major industries in the area, none of which are listed as having effluents or treatment plants. Figure 1 shows a portion of Lodi, bordering the Saddle River. The Saddle River is a tributary of the Passaic River, and is under the jurisdiction of the Passaic Valley Water Commission.

To obtain an accurate representation of a storm event, the area shown in Figure 1 was further divided into two, smaller sub-basins based on the existing network of storm sewers and topography. With regard to land use, these sub-basins may be characterized as follows:

Sub-Basin One	Sub-Basin Two
18 acres	25 acres
30% Residential	53% Residential
58% Commercial	42% Commercial
12% Industrial	5% Industrial

Although similar in size, Sub-Basin One has a considerably higher percentage of commercial and industrial land use than Sub-Basin Two, which may be characterized as primarily residential.

Sampling Urban Runoff

Centering our efforts on these two sub-basins, seven individual storm events were studied. Samples were taken manually at the sewer outfalls at 5 to 10-minute intervals over the complete hydrograph. This type of procedure required considerable on-station time--not only during runoff events, but also in anticipation of rainfall. Flow was determined by measuring the cross-sectional area and determining the velocity of runoff through the sewer. Stormwater samples were collected in acid-washed, two-liter polyethelene containers and acidified with nitric acid to a pH of 2. By measuring flow, and determining the concentrations of metals at these intervals, it was possible to measure directly the total "load" of heavy metals contributed to the receiving stream by runoff. In addition, an understanding of transport relationships may be more apparent by closer analysis of fluctuations of metals and suspended-solids concentrations observed with respect to time over the hydrograph.

Chemical Analysis

Suspended solids were determined in accordance with procedures outlined in Standard Methods. Total concentrations of lead, zinc, copper, chromium, and nickel were determined using a nitric:hydrochloric acid digestion procedure as outlined in Methods for Chemical Analysis of Waters and Wastes. All samples were analyzed directly after appropriate dilution using a Perkin Elmer 403 Atomic Absorption Spectrophotometer.

FIGURE 1

Saddle River at Lodi

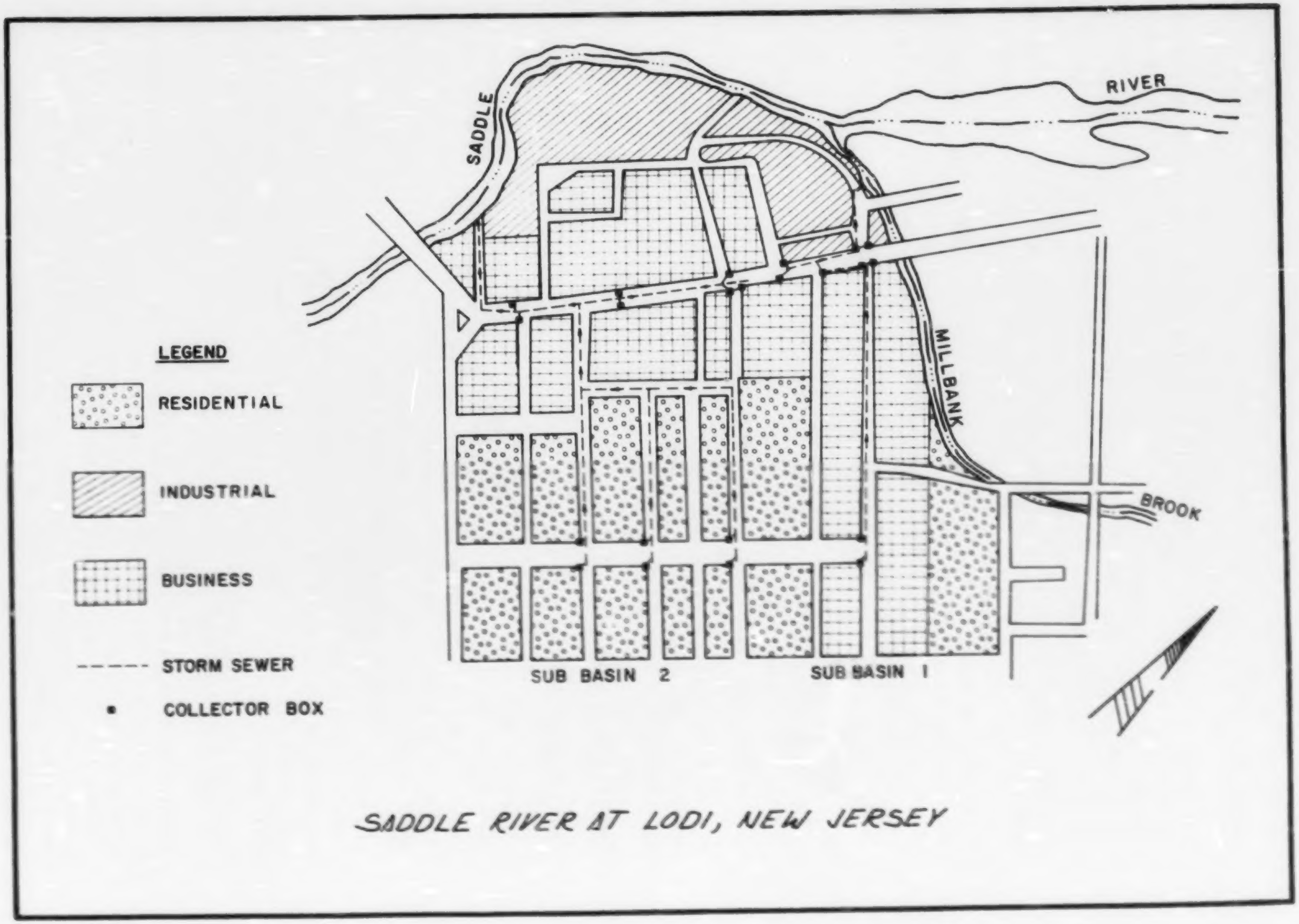

RIVER
SADDLE
MILLBANK
BROOK
LEGEND
RESIDENTIAL
INDUSTRIAL
BUSINESS
STORM SEWER
COLLECTOR BOX
SUB BASIN 2
SUB BASIN 1
SADDLE RIVER AT LODI, NEW JERSEY

FIGURE 2

Relationship of Lead in Storm Water Runoff to Flow and Suspended Solids. (July 24, 1974)

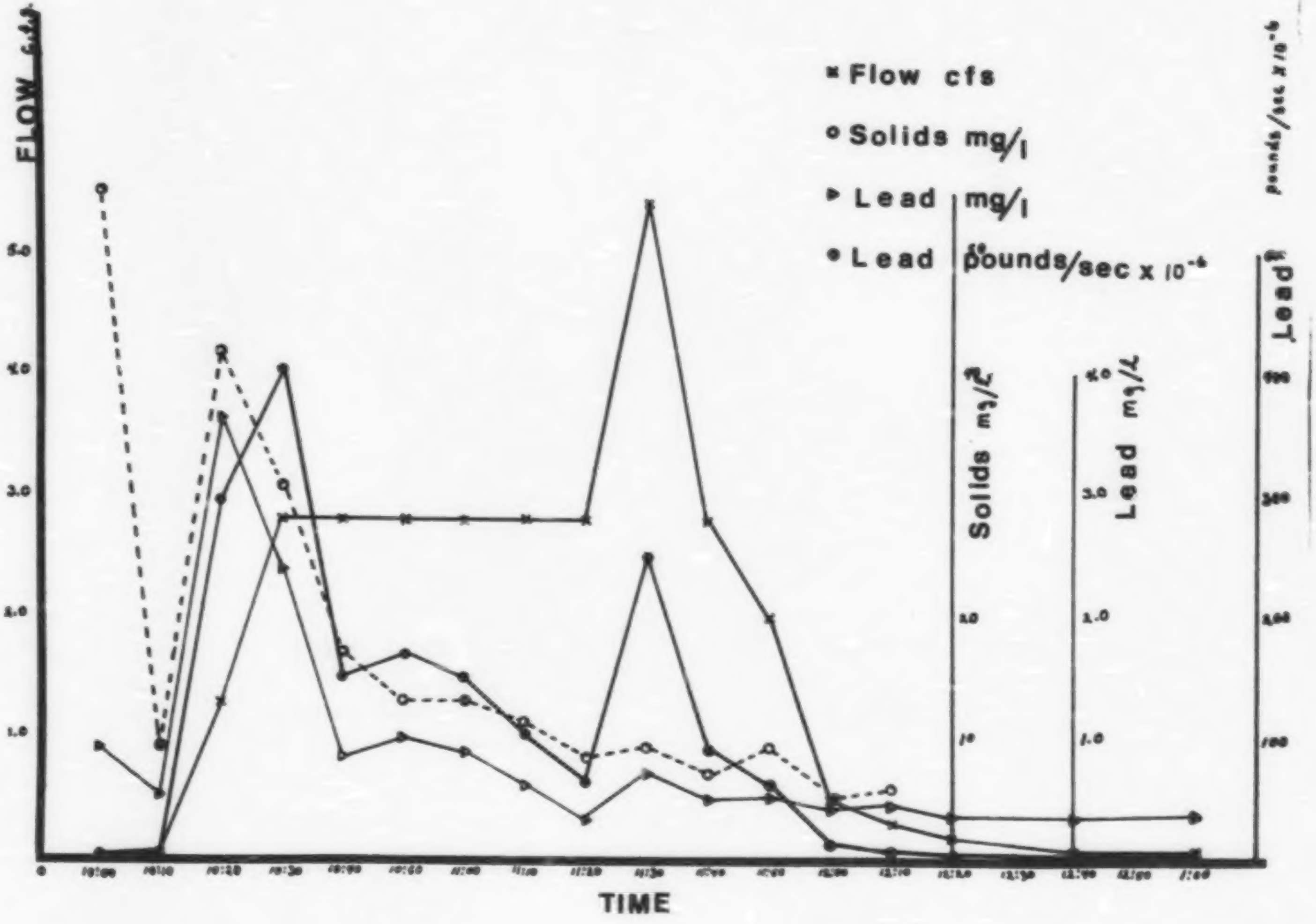
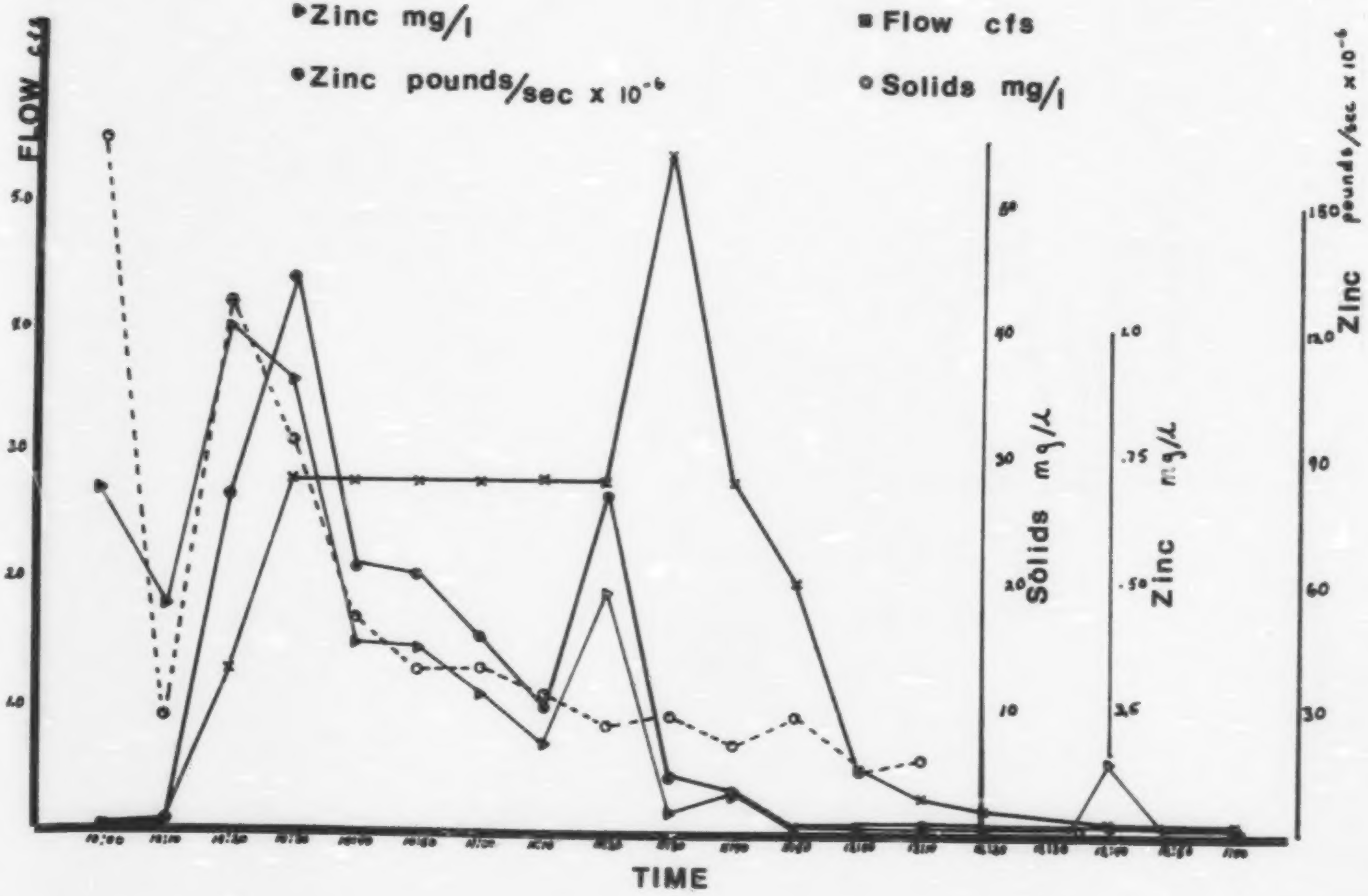


FIGURE 3



Relationship of Zinc in Storm Water Runoff to Flow and Suspended Solids. (July 24, 1974)

FIGURE 4

Relationship of Copper in Storm Water Runoff to Flow and Suspended Solids. (July 24, 1974)

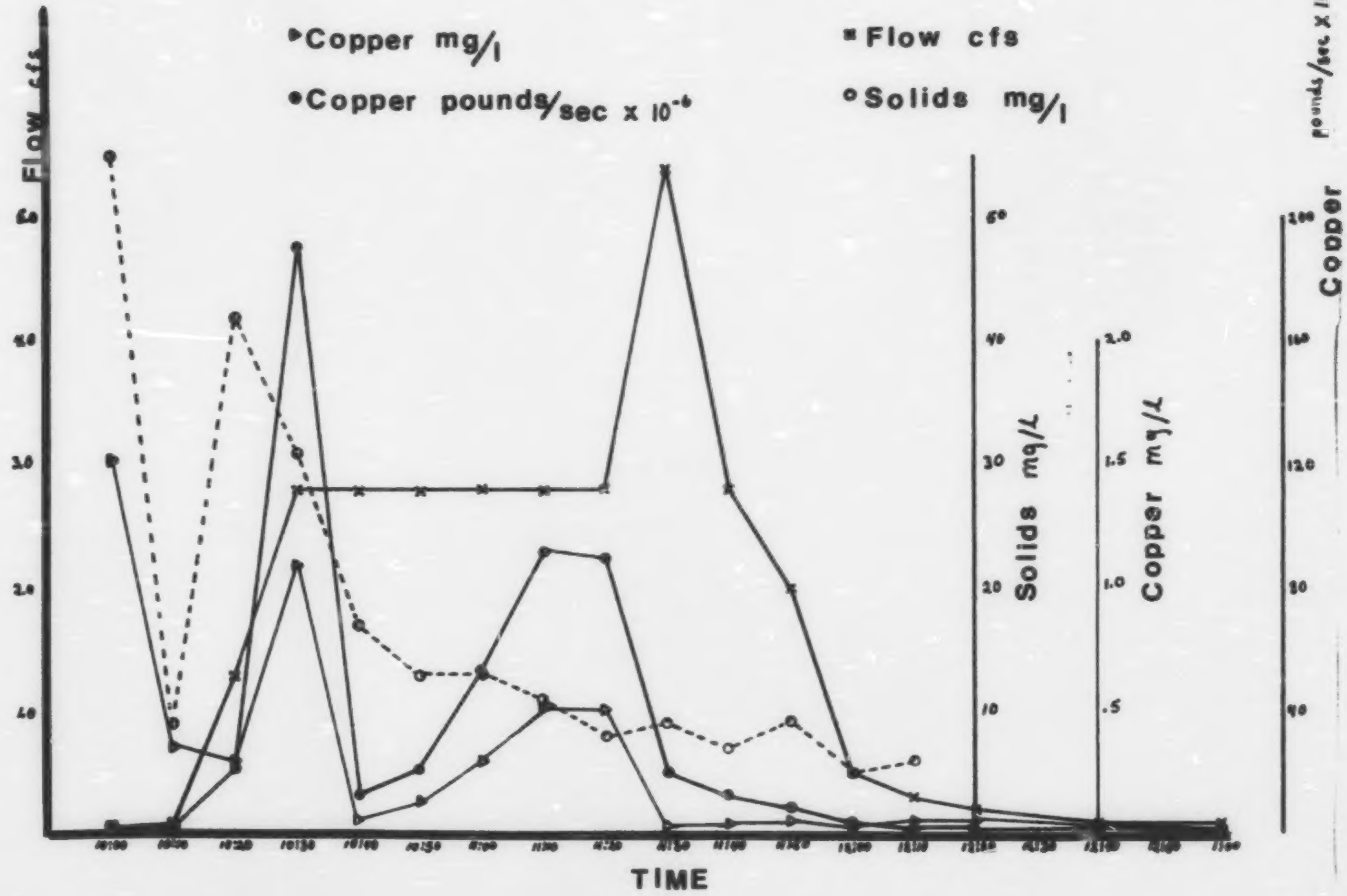


FIGURE 5

Relationship of Lead in Storm Water Runoff to Flow and Suspended Solids. (Aug. 7, 1974)

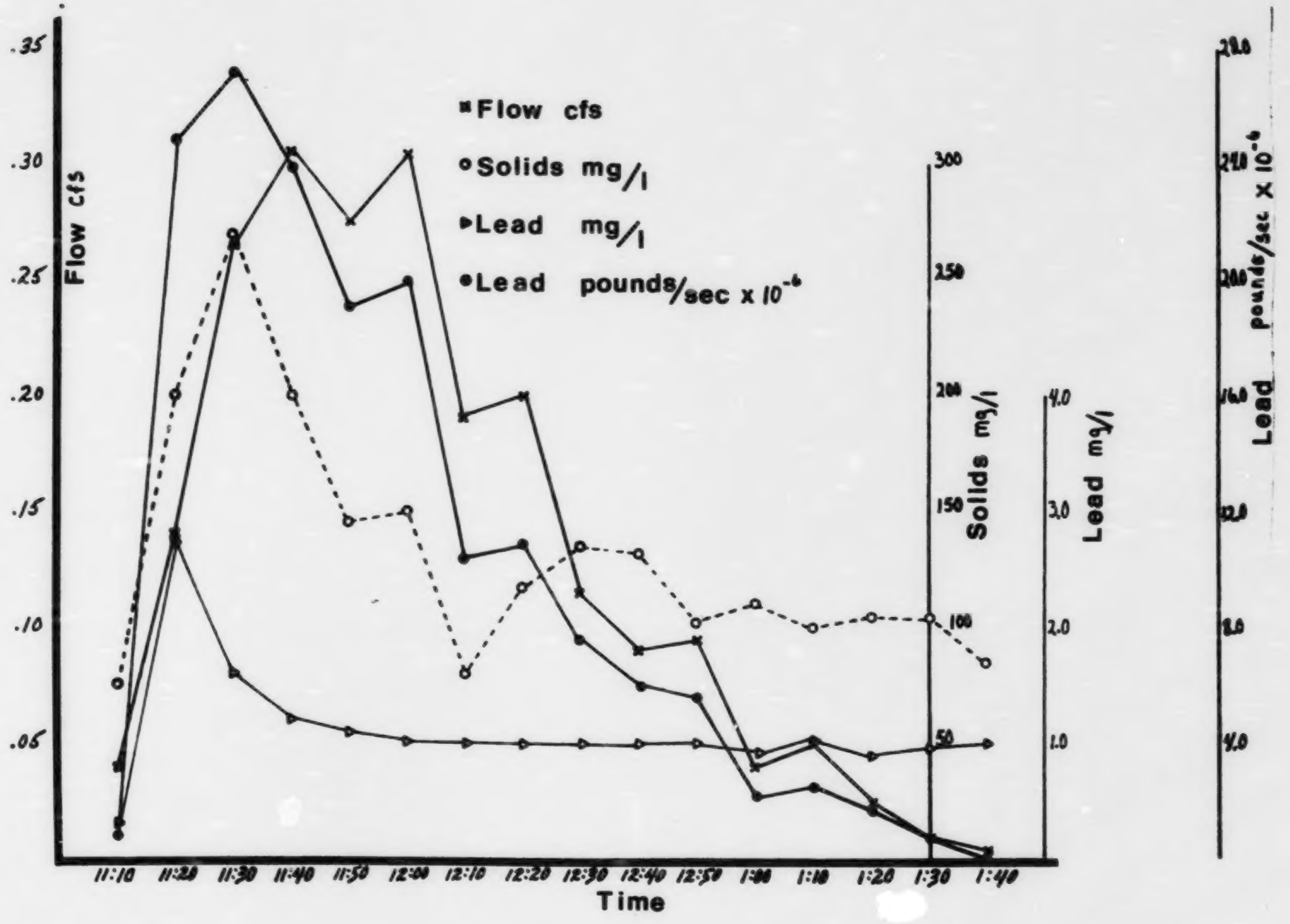


FIGURE 6

Relationship of Zinc in Storm Water Runoff to Flow and Suspended Solids. (Aug. 7, 1974)

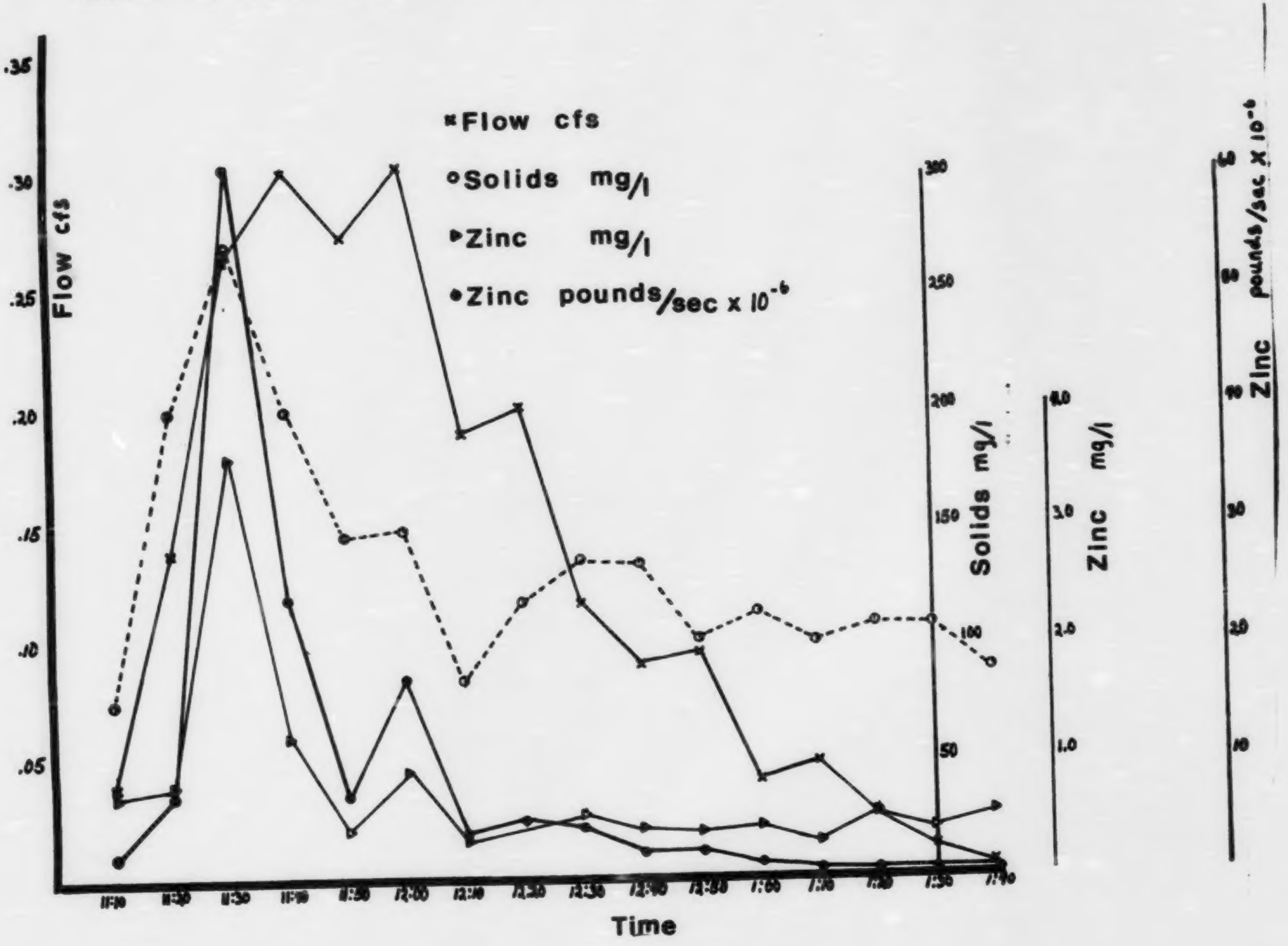


FIGURE 7

Relationship of Copper in Storm Water Runoff to Flow and Suspended Solids. (Aug. 7, 1974)

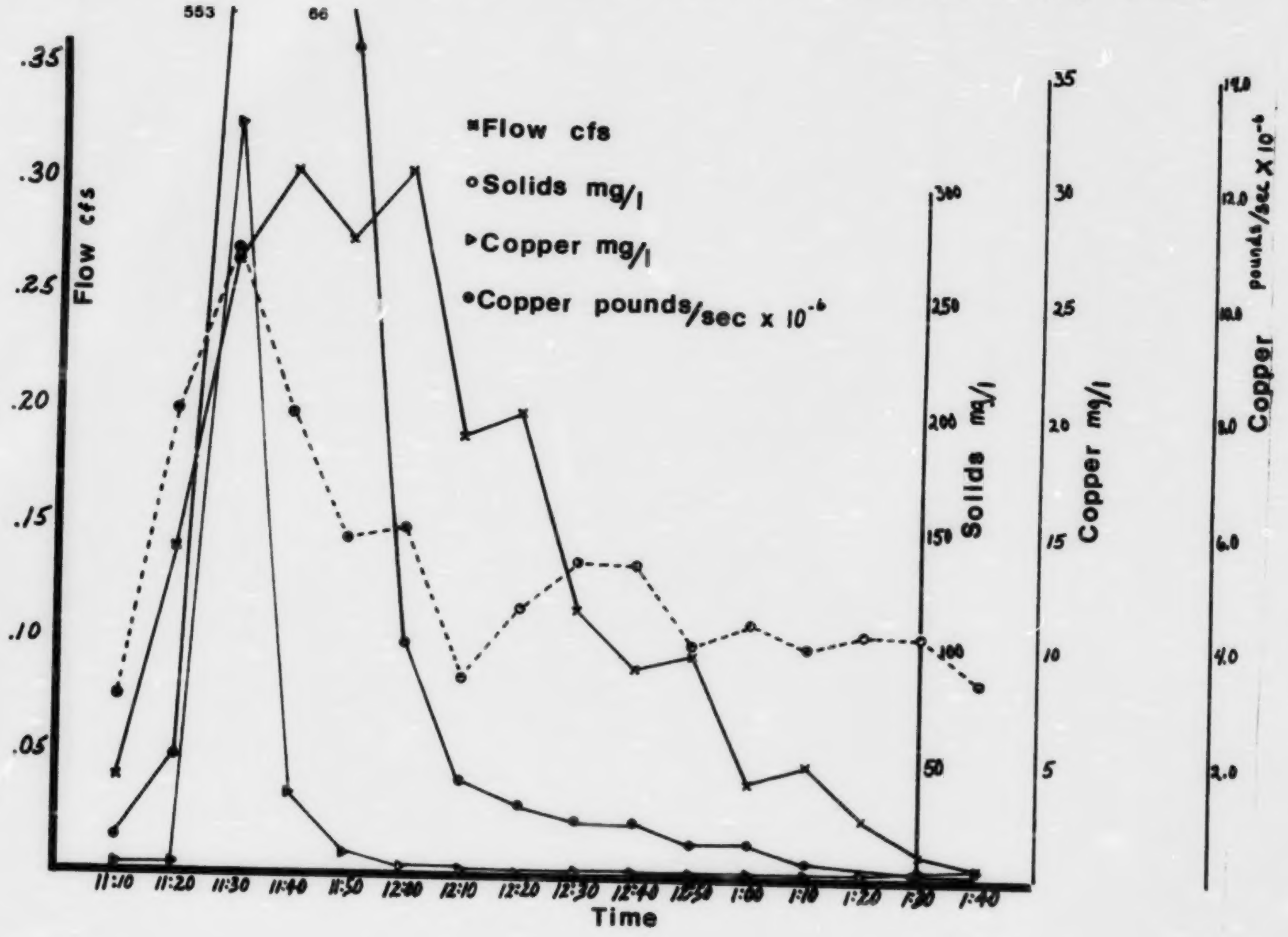


FIGURE 8

Relationship of Solid-fractions to Flow.

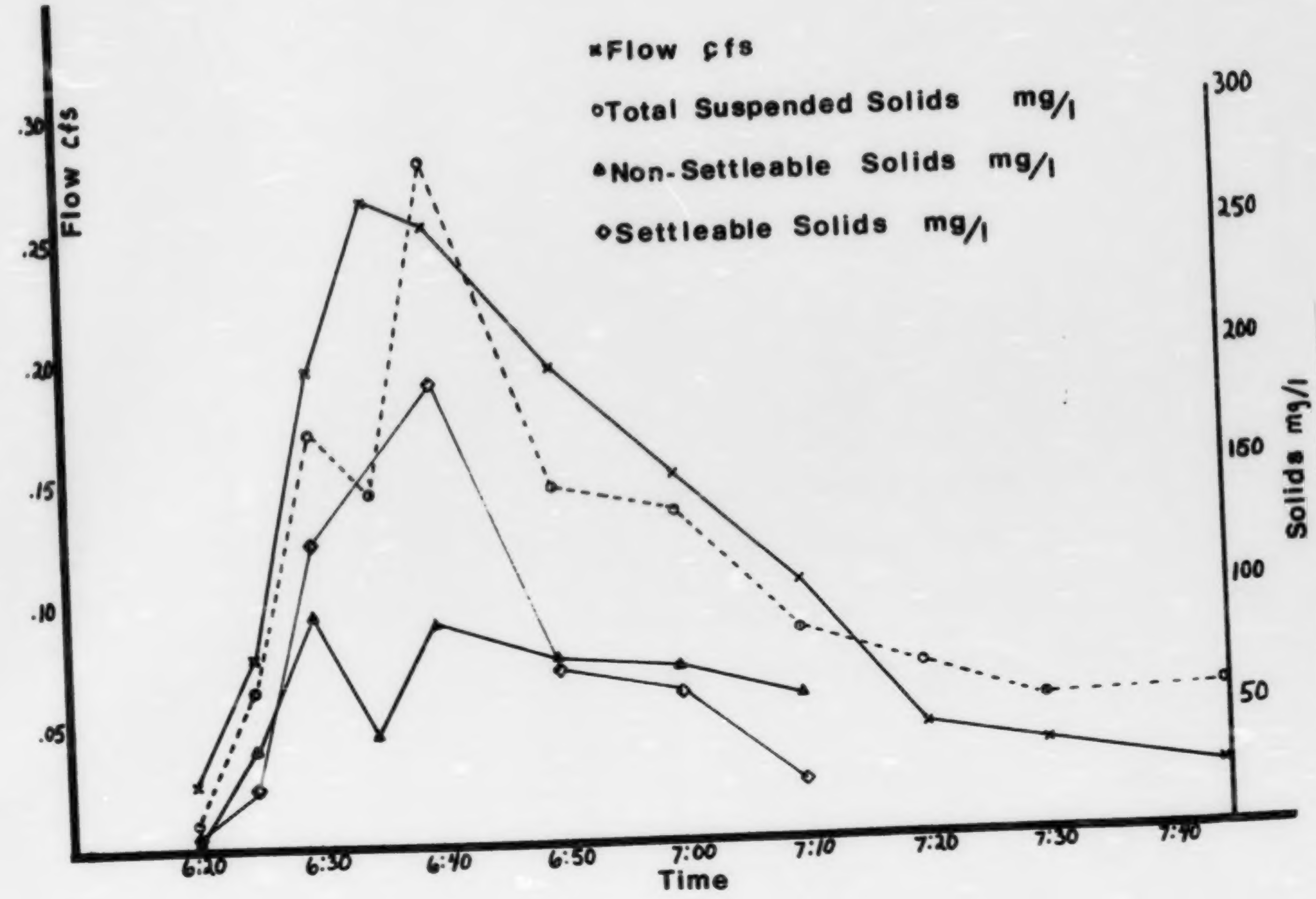


FIGURE 9

Relationship of Lead-fractions to Flow.

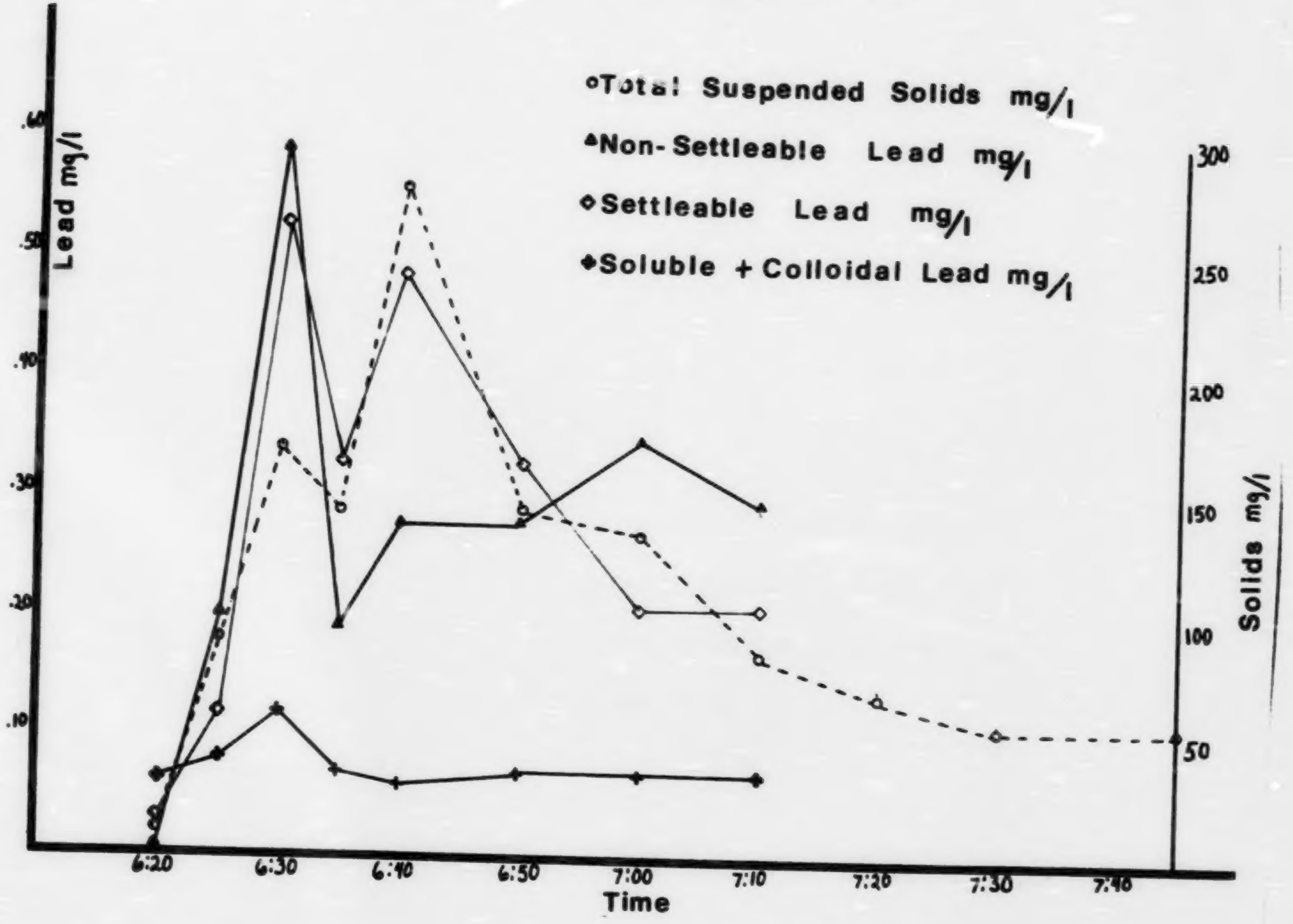


FIGURE 10

Relationship of Zinc-fractions to Flow.

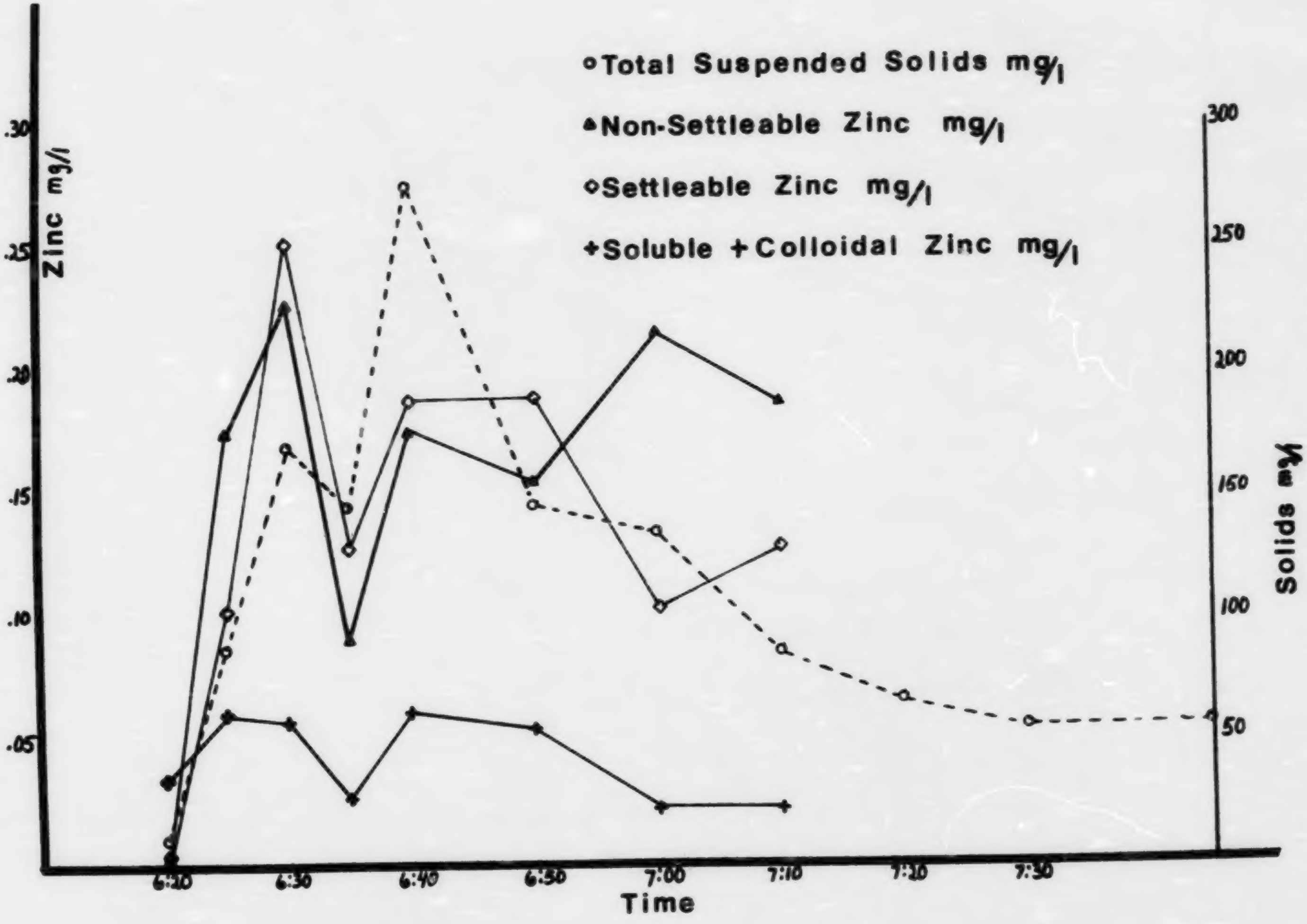
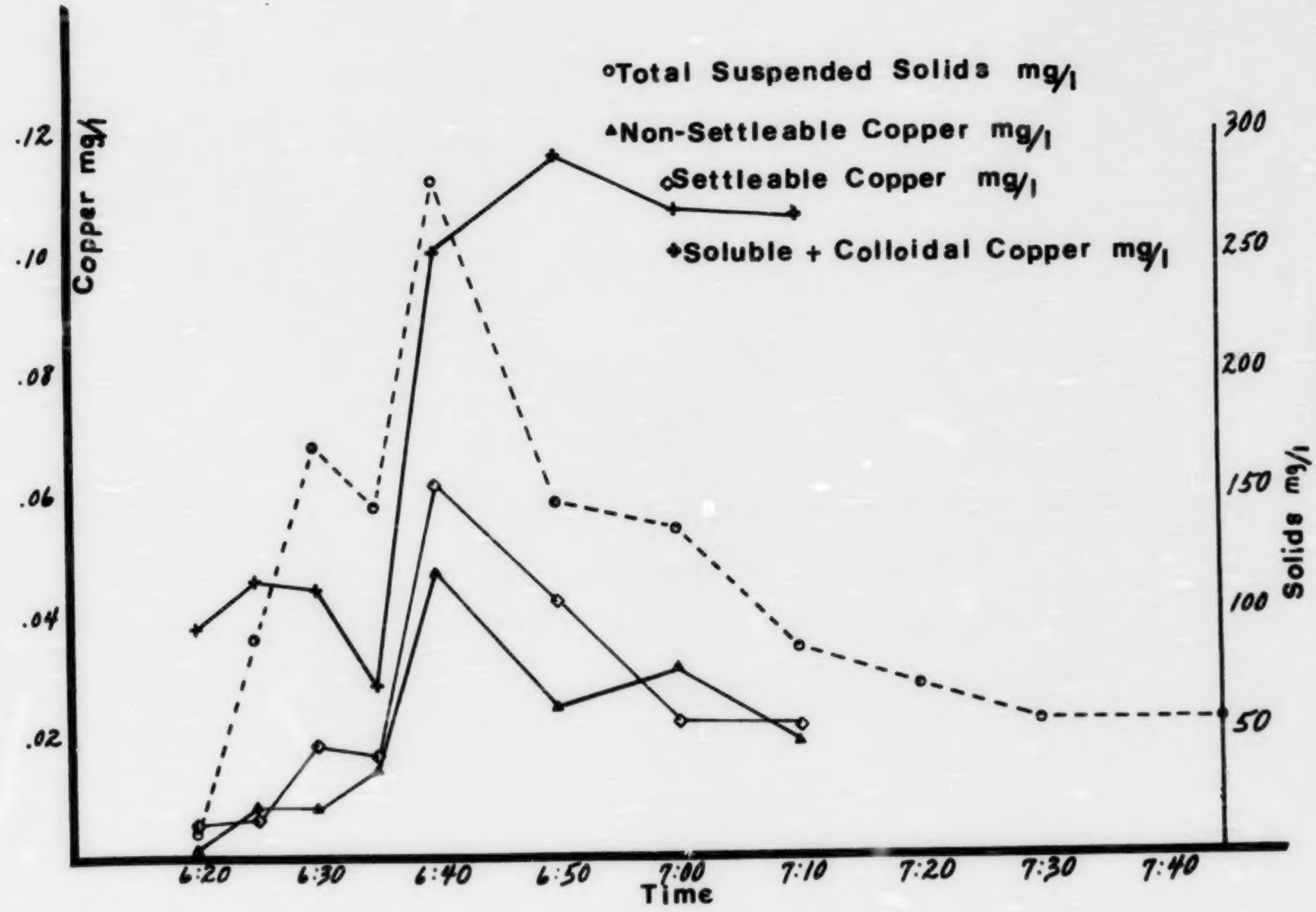


FIGURE 11

Relationship of Copper-fractions to Flow.



RESULTS AND DISCUSSION

Variations of Metals Concentrations

The variation in metals and suspended-solids concentrations were monitored at 5 to 10-minute intervals for seven runoff events over the study period. The first storms to be observed occurred on July 24, and August 7, 1974. Figures 2 through 7 illustrate the change in flow as well as concentration and load of the various constituents measured from the predominantly metal storm sewers. The interpretation of these diagrams is dependent upon the hydrologic conditions existing for these storms.

The storm event studied on July 24, 1974 [Figures 2,3,4] occurred after several weeks of dry weather and was the second in a series of light storms, the first storm resulting in .03 inches of rainfall. Total rainfall for the period sampled was .26 inches over a period of approximately 100 minutes. The rapid rise in flow during this storm is shown to have an initial flushing effect. Peak concentrations declined sharply until the point at which peak flow was achieved (11:30 pm). At this time smaller peaks in concentration were observed (.73 mg/l, .474 mg/l, and .525 mg/l, respectively). The initial peak in suspended solids was relatively low (42 mg/l) as compared to other storms studied, probably reflecting the flushing effect of the previous storm.

With respect to time, peak suspended-solids concentrations correlate with peak concentrations of lead and zinc, but are out of phase with respect to copper. It was noted that copper concentrations from the previous storm were considerably higher than those of the hydrograph under study.

The second storm studied (August 7, 1974) was considerably lighter in intensity than the first storm (.09 inches of rainfall in 135 minutes). The fact that it was the first storm of the day is reflected in the relatively high suspended-solids concentrations observed as compared to those of the July 24 storm. Figures 5, 6, and 7 demonstrate how lead, zinc, and copper varied with respect to time, flow, and suspended-solids concentrations. From these diagrams the following points may be noted. First, with respect to time, peak concentration of copper and zinc correlate with peak concentrations of suspended solids, while those of lead appear to be out of phase. Secondly, peak concentrations of copper reached 32.6 mg/l, roughly 30 times greater than those observed during the first storm. These observations most likely reflect industrial activity in the sub-basin studied, although other factors could be interacting. Similar studies conducted on areas with a higher percentage of residential land use resulting in copper concentrations considerably lower than those of the first two studies.

Fractionation Studies

In order to understand better the nature of heavy metals transport in urban runoff, the following studies were conducted in which the metals were partitioned on the basis of settleability and/or size.

As in other studies, samples were collected at 5 to 10-minute intervals over a storm hydrograph. At each interval two samples were collected, one of which was acidified. The remaining sample was allowed to settle for several hours, after which the supernatant was siphoned off. This supernatant fraction was either analyzed directly to obtain a "non-settleable" fraction, or further fractionated by filtration (pore size approximately .75 to 1.25 μ) to obtain a fraction containing metals in both the soluble and colloidal phases. These fractions will be reported herein as "Settleable Solids" (total concentration - "Non-Settleable"); "Non-Settleable Solids" (Non-Settleable Fraction - "Soluble + Colloidal"), and "Soluble Plus Colloidal" (as described above). As an illustration of one such study, see Figures 8, 9, 10, and 11.

Figure 8 illustrates the relationship of the various solid fractions to flow. Several observations concerning this data may be made. First one notes what appears to be a high degree of correlation between flow and "Settleable Solids." Particle settleability is a function of specific gravity and particle diameter. In all probability, the observed correlation reflects the relationship between these two factors and scour velocity. As one would expect, "Non-Settleable Solids," which have a lower specific gravity, were flushed during the initial increase in flow.

The distribution of the various lead fractions may be seen in Figure 9. It is significant to note that the majority of lead is found in the "Non-Settleable" and "Settleable" solids fractions (38.8 percent and 42.5 percent, respectively), while very little is found in the "Soluble plus Colloidal" fractions. As in the solids fractions, "Non-Settleable" lead concentrations were highest during the initial stages of flow. "Settleable" lead concentrations also reached peak values during the initial stages of flow. However, in general, "Settleable" lead tended to follow total suspended-solids concentrations over the hydrograph.

A comparison of the different zinc fractions [Figure 10] shows that the majority of this metal also is contained in the "Non-Settleable" and "Settleable" solids fractions (44.5 percent and 35.8 percent, respectively). This information is in complete agreement with observations made in prior studies. In general, both "Settleable" and "Non-Settleable" fractions tend to follow total suspended-solids concentrations over the hydrograph.

Copper, in contrast to lead and zinc, was found predominantly in the "Soluble plus Colloidal" fraction (66 percent) [Figure 11]. However, as in some other studies, total peak concentrations of copper were found to occur at the same time as peak concentrations in suspended solids. "Settleable" and "Non-Settleable" fractions of copper, in general, were observed to follow total suspended-solids concentrations over the hydrograph.

Nickel and chromium, although in considerably lower concentrations, also tend to predominate in the "Soluble plus Colloidal" fraction (74.3 percent and 76.4 percent, respectively).

On the basis of percent distribution, "Settleable constituents were observed to predominate over the ascending limb of the hydropgraph while "Non-Settleable" constituents began to predominate in the latter stages of the descending limb. In this study, the change in distribution was observed at about 6:50. These observations probably can be attributed to scour, which is directly related to flow.

Heavy Metal Loadings

As mentioned previously, one of the objectives of this research was to evaluate the

TABLE 1

Results of Total Loadings for Two Storms Studied On Each Sub-Basin
(Figured in pounds/storm)

Sub-Basin One (roughly 18 acres)

Storm 1: Total rainfall: .26 inches in 100 minutes. This rainstorm preceeded by smaller storms. Antecedent dry period 4 days.

Pb	Zn	Cu	Cr	Ni	Sus. Solids
1.08	.3009	.3065	.032	.0275	14.12

Storm 2: Total rainfall .09 inches in 135 minutes. Antecedent dry period 2 days.

Pb	Zn	Cu	Cr	Ni	Sus. Solids
.093	.0813	.4107	.002	.0099	13.64

Sub-Basin Two (roughly 25 acres)

Storm 1: Total rainfall .04 inches in 75 minutes. Antecedent dry period 6 days.

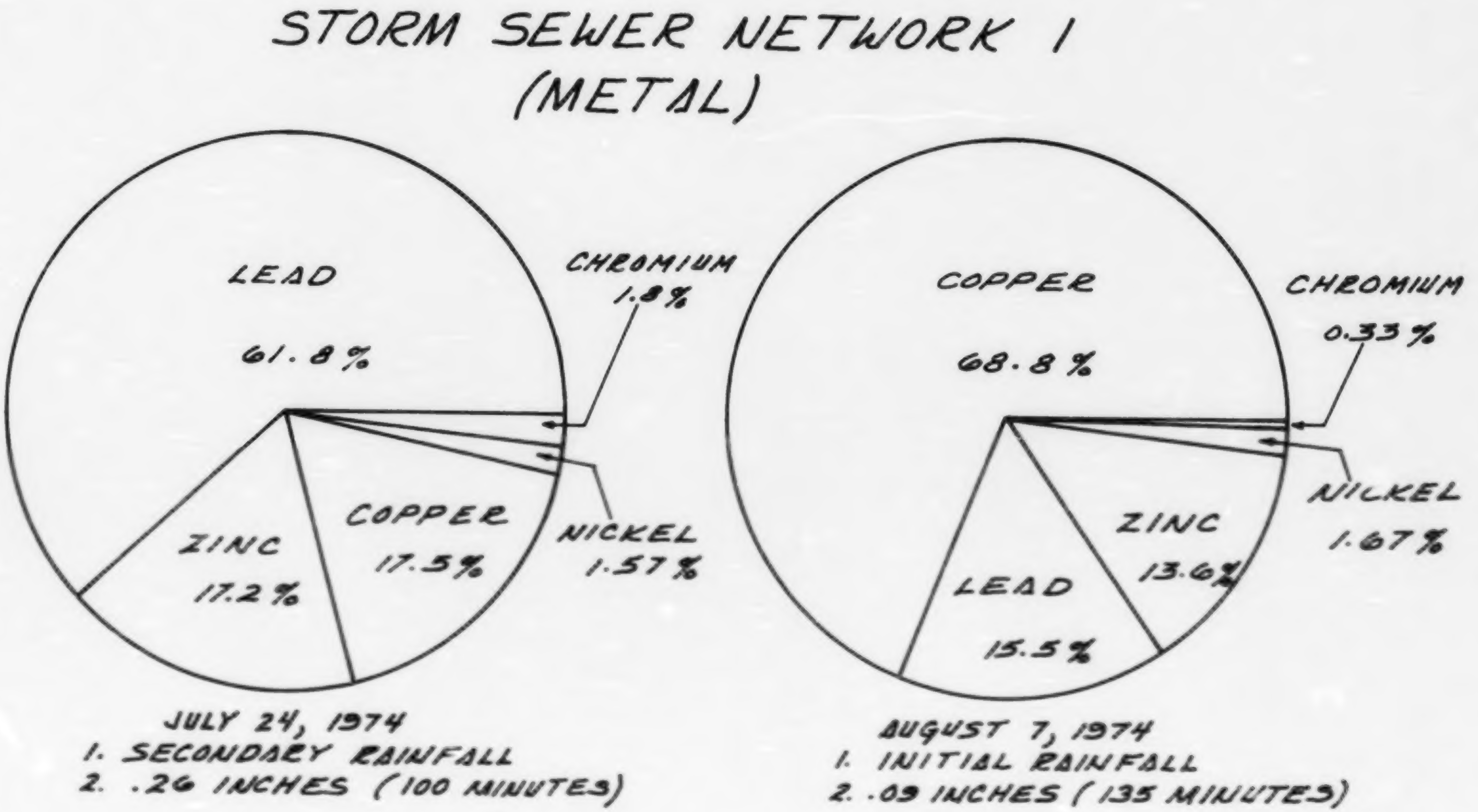
Pb	Zn	Cu	Cr	Ni	Sus. Solids
.025	.021	.003	.0012	.0053	10.64

Storm 2: Total rainfall .19 inches in 145 minutes. Antecedent dry period 5 days.

Pb	Zn	Cu	Cr	Ni	Sus. Solids
.0763	.0578	.0095	.0026	.0031	14.94

FIGURE 12

Relative Contributions of Five Metals Representing the Total Loadings of Two Individual Storm Events

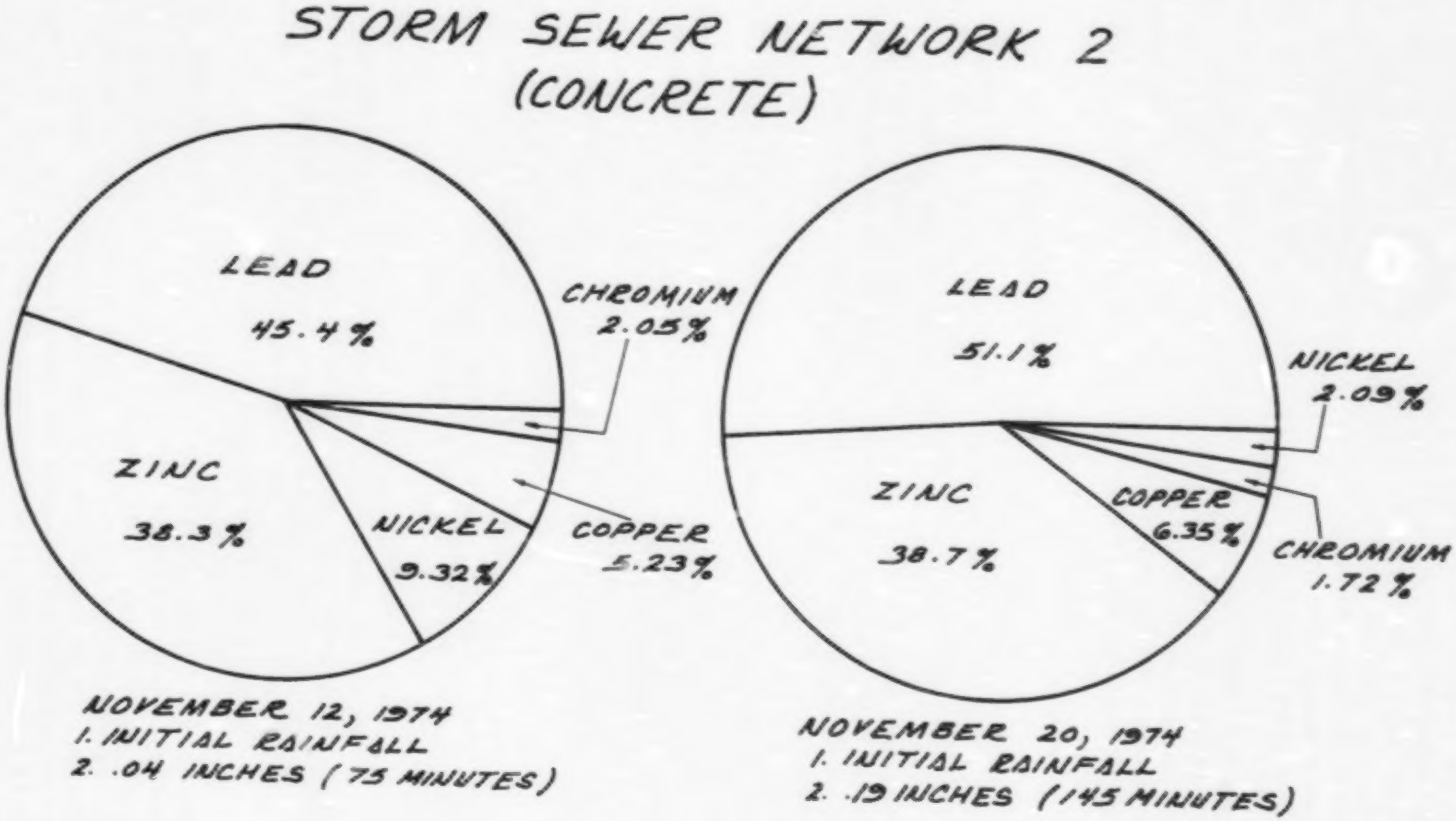


JULY 24, 1974
1. SECONDARY RAINFALL
2. .26 INCHES (100 MINUTES)

AUGUST 7, 1974
1. INITIAL RAINFALL
2. .09 INCHES (135 MINUTES)

FIGURE 13

Relative Contributions of Five Metals Representing the Total Loadings of Two Individual Storm Events



total loadings of heavy metals being contributed during individual storm events. These contributions were found to vary widely between storms, probably as a result of such factors as: (1) the length of antecedent dry period; (2) intensity and duration of the storm event; (3) size of the drainage basin; (4) land use; (5) composition of the storm sewer network, and (6) general effectiveness of waste removal in the streets. Storms studied varied in intensity from .04 inches of rainfall in 75 minutes to .26 inches of rainfall in 100 minutes [see Table 1].

In general, lead, zinc, and sometimes copper were the major contributors of heavy metals in the storm-water runoff examined. Together, these accounted for approximately 90 to 98 percent of the total metals observed. Of this, lead and zinc account for approximately 80 percent. Chromium and nickel usually were found in considerably smaller quantities, as is shown in Figures 12 and 13.

SUMMARY AND CONCLUSIONS

On the basis of studies conducted over several storm hydrographs in Lodi, New Jersey, it appears that the major contributors of heavy metals in urban runoff are lead, zinc, and copper. Furthermore, these generally appear in greatest concentration shortly after the initiation of runoff, usually within the first 30 minutes, thus giving a first-flush effect. Peak concentrations, with the exception of lead, extend over a short period of time, usually in the range of 10 to 20 minutes.

Closer analysis of the metals studied in this storm-water runoff show the following:

1. Peak concentrations of lead, in all cases, were found to occur prior to, or at the same time, as peak concentrations in suspended solids. In addition, roughly 80 percent of the lead concentrations observed were found to be associated with the "Non-Settleable" and "Settleable" solids of storm-water runoff.

2. In all of the hydrographs studied, the peak concentrations of zinc coincided in time exactly with peak concentrations of suspended solids. As with lead, these concentrations were associated primarily with the solids fractions.

3. Copper concentrations, originating from the predominantly commerical and industrial sub-basin studied (Sub-Basin One), were found to be considerably higher than those originating from the predominantly residential sub-basin (Sub-Basin Two). In contrast to lead and zinc, these concentrations tended to predominate in the "Soluble Plus Colloidal" fraction.

Comparison of heavy-metals concentrations in urban runoff to those in secondary treatment plant effluents, as reported by the Interstate Sanitation Commission [Table [illegible]], indicates that the storm-water runoff contributes a substantially greater portion of metals to a receiving body than a normal sewage-treatment plant on a unit volume basis. The environmental impact of this is unclear. On the basis of a flow-weighted mean, lead is the only metal studied that exceeds the Permissible Criteria for Public Water Supplies. Possibly more important, however, are peak concentrations of heavy metals which are reached during the ini-

TABLE 2

A Comparison of Heavy Metals from Various Sources with Toxicological Data and Water Quality Criteria

	Lead mg/l	Zinc mg/l	Copper mg/l	Chromium mg/l	Nickel mg/l
Urban Runoff Lodi, N.J.	0.90	0.62	0.15	0.03	0.08
Secondary Treat. Plant Effluents	0.20	0.08	0.05	0.05	0.10
Ratio $\frac{\text{Runoff}}{\text{Sewage}}$	4.5	7.75	3.0	0.60	0.80
Rainfall (6)	0.034	0.107	0.021	--	0.043
Toxic Concentrations to assorted fish species (Schroeder)	2.0	2.0	0.10	--	4.0
Surface Water Qual. Criteria (Permissible)	0.05	5.0	1.0	0.05	none

tial phases of storm runoff. Quite frequently these concentrations exceed those which have been found to be toxic to various species of freshwater organisms [McKee and Wolf, 1963]. The effect of these "slug" concentrations has not yet been determined.

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STREAM REHABILITATION THROUGH CONTROL OF NON-POINT SOURCES OF ACID MINE DRAINAGE

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ABSTRACT

Experimental studies showed the mechanisms of stream rehabilitation were related to improvement of physical-chemical conditions degraded by acid mine drainage, combined with reintroduction of aquatic organisms to the restored aquatic habitats. When physical, chemical, and biological conditions can be improved by seasonal variation in stream conditions or changes in mining practice, implementation of certain remedial actions can affect transformation of non-point sources of acid mine drainage to point sources, facilitating wastewater treatment to eliminate acid mine drainage at or near its source and resulting in restoration or rehabilitation of the stream system.

INTRODUCTION

Mine discharges may adversely affect both surface and ground water resources, altering the quality of receiving waters and producing severe environmental impact. The recent expansion of the mining industry, especially surface mining, coupled with increased environmental concern has made maintenance of high-quality surface waters an essential part of mineral extraction. For the most part, discharges from mining operations are considered a non-point source of pollution, even though the nature of a watershed will limit mine drainage to stream channels where highly concentrated mine drainage can produce severe environmental impact similar to that caused by source discharges. The essential element of control of this type of pollutional discharge is recognition that the overall environmental impact of the discharge can be reduced by manipulating the watershed which controls the discharge (e.g., volume, concentration, hydrograph characteristics, etc.). The approach to system manipulation we have used recognizes that streams and rivers are an integral part of the total mining effort, including reclamation and revegetation of watershed lands. This approach encourages the enlightened multiple use of the stream system based on an understanding of the interrelationships among the stream's physical, chemical, and biological elements.

Since streams are the major emphasis of discussion in this paper, a review of the essential elements of stream function is appropriate before discussing techniques that can be used to reclaim and rehabilitate damaged stream systems.

STREAM FUNCTIONAL ELEMENTS

Physical

Physical data (flow, temperature, etc.) is of major importance in determining the impact any discharge will have on the stream system. Key among the various physical parameters is stream flow or discharge. Discharge values reflect the hydrologic response of the stream to changes occurring throughout the watershed, providing a single parameter which relates changes in the physical-chemical conditions to changes in stream biota. Furthermore, information on discharge quality and quantity, combined with specific wastewater volumes and characteristics, may provide an accurate estimate of the overall impact the discharge may have on the environment.

Chemical

Stream chemical conditions are dependent not only on the various discharges which contribute to total stream flow, but also on watershed parameters such as soil type, geology, or vegetative composition. In streams where mine discharges, especially acid mine drainage (AMD), contribute significantly to stream flow, water quality may vary considerably with seasonal conditions which affect not only dilution water composition, but also waste-discharge composition and concentration. These relationships will be discussed in detail later in this paper.

Biological

The biological system is the key element in judging overall stream conditions. The biota of a stream subjected to pollutional stress may respond in predictable ways, based on interactions between physical-chemical and biological components of the stream. The recovery or rehabilitation of the stream biota is dependent on the type, intensity (e.g., concentration), and duration of the stress which caused damage. For example, a short-duration stress such as a spill of toxic material will generally be quickly moderated by continued stream flow (dilution) removing the stress near the point of discharge, and reducing the stress downstream by dilution and diffusion. Long-duration stress--typical of AMD--caused by a point source (or non-point sources where the location of the discharges is limited to one area) continues to affect the area near the discharge location, and in most situations is moderated as a function of distance from the discharge source. Recovery of the biological system is related to physical (e.g., discharge, watershed morphology and/or geology), chemical (e.g., water quality or characteristics), and biological (e.g., presence and abundance of biota, toxicity, or sources of recolonizing organisms) components of the stream tying all elements of the stream functional system together.

TABLE 1

Criteria for Determining Acid Mine Drainage

pH	less than 6.0
Acidity	greater than 3 mg/l
Alkalinity	normally 0
Alkalinity/Acidity	less than 1.0
Fe	greater than 0.5 mg/l
SO_4	greater than 250 mg/l
Total Suspended Solids	greater than 250 mg/l
Total Dissolved Solids	greater than 500 mg/l
Total Hardness	greater than 250 mg/l

ACID MINE DRAINAGE CHARACTERISTICS

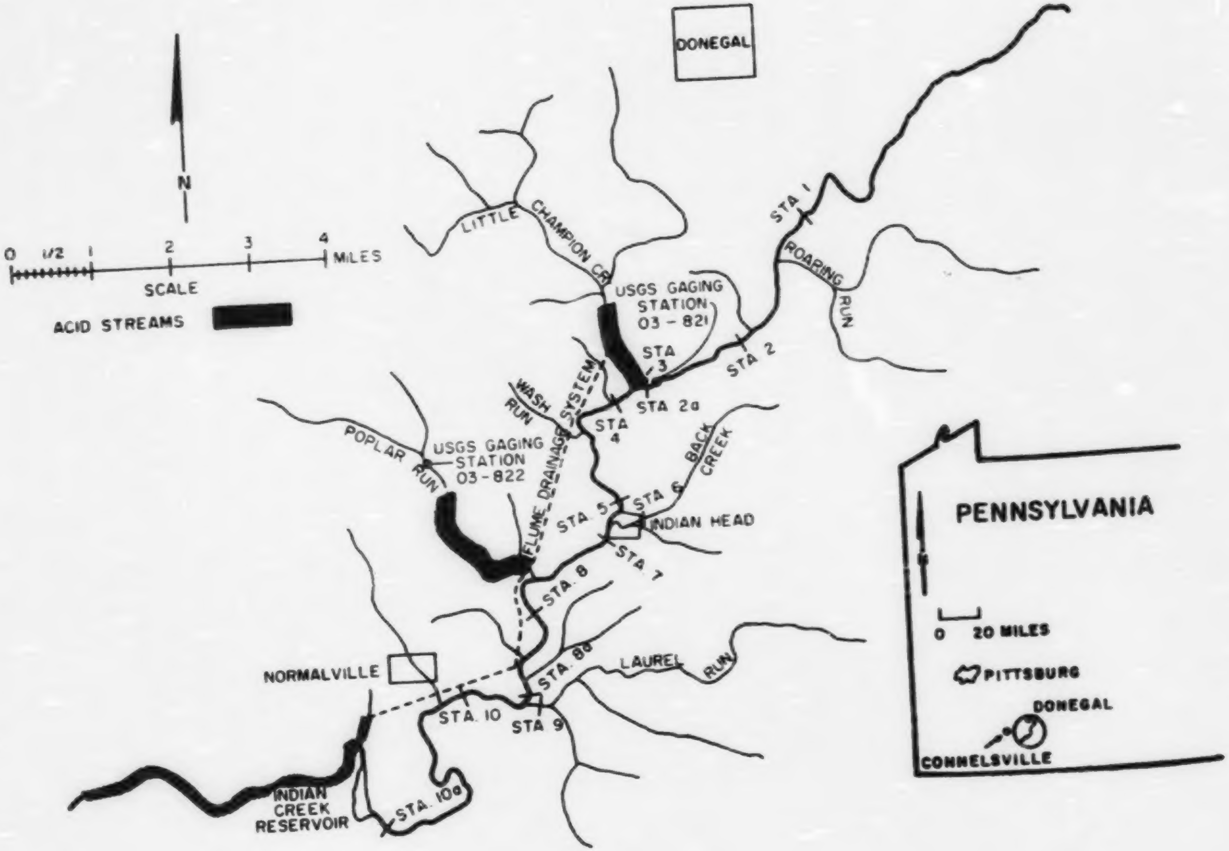
Acid mine drainage originates from the oxidation of sulfide minerals. By far the most common acid producing sulfide mineral is iron sulfide, but other sulfide minerals (Cu_2S, ZnS, or PbS) may be found associated with acid-forming deposits. When the sulfide mineral is exposed to oxygen in a moist environment, the sulfide oxidizes, producing metal salts and sulfuric acid. The end product of this reaction can be described as a solution of mixed salts containing a variety of elements, and possibly charged with carbon dioxide, depending on the acid source. The origin of acid discharge, whether from underground or surface sources, and the mineralogy of the coal seam play an important role in the formation, intensity, and eventual impact of the mine drainage. If the alkalinity of the mine water is not sufficient to neutralize the ferrous sulfate produced by oxidation, the resulting mine drainage will become strongly acidic upon aeration (e.g., after entering a stream). Addition of dilution water without a high alkalinity promotes the hydrolysis of the ferrous sulfate, releasing more hydrogen ions and further depressing the pH.

The quality of water affected by acid mine drainage is variable, but general criteria for the identification of streams with major mine drainage influence are given in Table 1. Due to the low pH, the dissolved solids loading may contain significant quantities of iron, aluminum, and other heavy metals, depending on the mineralogical composition of the ore deposit and associated strata. The most useful indicator of the presence and concentration of acid mine drainage is sulfate. Calcium sulfate, the most common neutralization product, is soluble at concentrations usually encountered in receiving streams. The other minerals in acid mine drainage tend to precipitate or plate out of solution and are difficult to analyze reliably as the pH and alkalinity of the receiving water change. Because sulfates usually are present in low concentration in Appalachian waters (less than 20 ppm) and are found in high concentrations in acid mine drainage, the presence of sulfate gives an accurate indication of the presence of mine drainage. In addition, one molecular weight of sulfate is formed for each molecular weight of sulfuric acid, allowing interpretation of total acid loading of streams from increased sulfate concentration over background levels.

FIGURE 1

Indian Creek Drainage Map

FAYETTE COUNTY, PA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POPLAR RUN
USGS GAGING STATION 03-822
FLUME DRAINAGE
STA 7
STA 8
STA 8a
LAUREL RUN
NORMALVILLE
STA 10
STA 9
INDIAN CREEK RESERVOIR
STA 10a
PENNSYLVANIA
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PITTSBURG
DONEGAL
CONNELSVILLE

STREAM SYSTEM RECOVERY FROM ACID MINE DRAINAGE

The experimental watershed selected for detailed physical, chemical, and biological studies was the Indian Creek Basin, Fayette County, Pennsylvania [Figure 1]. Fifteen stations in the Indian Creek Basin were sampled for both biological and physical-chemical information. These stations included both mainstem and tributary stations, and were established to provide reference areas, areas severely affected by mine drainage, and areas of recovery from mine-drainage stress. Thirteen biological samples were made during the two-year period of October, 1970 through October, 1972. In addition to water-quality analyses performed with biological samples, supplementary water-quality analyses were made during specific events (high flows), and to identify various watershed characteristics. Physical sampling included plane table surveys of each biological sampling station, and geological reconnaissance of the entire watershed. Details of sampling procedures and results are available in Herricks and Cairns [1974].

Physical-Chemical Studies

Physical-chemical studies included both field sampling and computer simulation. Water-quality parameters were monitored on a regular basis. These included analysis for pH, sulfate, heavy metals, total acidity, alkalinity and hardness, dissolved oxygen, and conductance. A portion of this data--mean and standard deviations for total acidity and alkalinity, pH, and sulfate--are contained in Table 2. The remainder of the water-quality data is available in Herricks and Cairns [1974].

Information on discharge quality and quantity, combined with acid mine drainage volumes and characteristics, may provide an accurate estimate of the overall impact on the environment. Unfortunately, stream flows were available for only one sub-basin in the Indian Creek watershed. To overcome the difficulty of making accurate interpretations without discharge data, hydrologic simulation techniques were used both to generate flow data for all major sub-basins which had no historical stream flow records, and to provide basic data to allow development of water-quality models to better define the impact of mine drainage on Indian Creek [Herricks, et al., in press].

A summary of water-quality analyses during the two-year sampling effort can be seen in Table 2. At Station 1, water quality was good. Alkalinity values were consistently high, while acidity values were consistently low. Sulfate values had a small range, with the average value below 15 ppm. Station 2 showed similar results, but Station 2a had higher mean values for acidity and sulfate, and a greater range for all four parameters. Station 3, Champion Run, had consistently high acidity and sulfate, no alkalinity, and low pH. The ranges for all parameters were large, indicating variable but poor water quality. This may be attributed to variable discharge/acid mine drainage volume relationships in Champion Run. Station 4 was unmixed. The right-bank samples show lower mean acidity and sulfate values, and higher mean alkalinity when compared with the left bank. The range of values recorded for both right and left bank are large, indicating variable water quality. This variability may also be attributed to changes in discharge relationships between the upstream portion of Indian Creek and Champion Run. Station 5 had a higher mean acidity and sulfate values, and a lower mean alkalinity when compared with Stations 1 or 2. The water quality of the mainstem of Indian Creek improved considerably after the confluence with unpolluted tributaries. There were no major tributaries between Station 4 and Station 5. Station

TABLE 2

Mean and Standard Deviation Values for Four Water-Quality Parameters
Indian Creek, Fayette County, Pennsylvania

STATION	1	2	2a	3	4R	4L	5	6	7	8	9	10
pH												
mean	7.2	6.9	6.4	4.0	6.0	5.8	6.6	7.1	6.9	6.6	7.1	7.0
S.D.	0.2	0.4	2.2	0.4	1.6	0.6	0.3	0.2	0.3	0.9	0.2	0.4
total acid												
mean	3.8	2.5	4.3	50.7	6.1	12.0	4.9	2.8	4.3	2.6	2.8	3.7
S.D.	3.6	2.3	3.4	24.3	5.8	8.9	3.6	3.0	4.0	2.7	2.8	2.7
total alkalinity												
mean	23.6	21.3	22.3	0.0	16.5	6.5	10.4	21.0	12.7	16.3	16.8	11.5
S.D.	7.6	7.6	10.0	0.0	7.5	3.6	6.1	8.1	5.1	3.2	6.3	5.3
sulfate												
mean	13.2	15.8	20.4	128.2	27.6	49.0	33.0	10.0	35.5	40.9	8.9	29.9
S.D.	2.0	4.7	6.2	82.0	11.3	16.8	11.4	3.1	9.4	15.3	1.4	7.1

6 and Station 9 were unpolluted tributaries. All four parameters listed in Table 1 were similar to mean and range values recorded for Station 1. The addition of these unpolluted tributary waters improved the mainstem water quality not only by dilution, but also by chemical reaction. At Station 7, mean alkalinity values increased while mean acidity values remained similar when compared with Station 5. Water quality at Station 8 showed further improvement. Poplar Run was another acid tributary. Sulfate values were consistently high, as was mean acidity; mean alkalinity was low. By Station 10, mean sulfate values were lower. The range of acidity values for Station 10 was less than Station 7, and the mean alkalinity value, although lower than Station 8, had little variability.

In summary, the water quality in the mainstem of Indian Creek illustrated the overall effect of a severe mine drainage influence. The upstream stations maintained good water quality throughout the study period. Although mine drainage from Champion Run was poorly mixed at Station 4, water quality was, on the average. reduced at this station. With complete mixing of the acid mine drainage and no tributary influence, Station 5 had the poorest water quality when all parameters are considered. Although sulfate concentrations increased throughout the stream, the addition of unpolluted tributary water improved water quality. The water quality at Station 7 was variable; this variability decreased by Station 10, where overall water quality was improved.

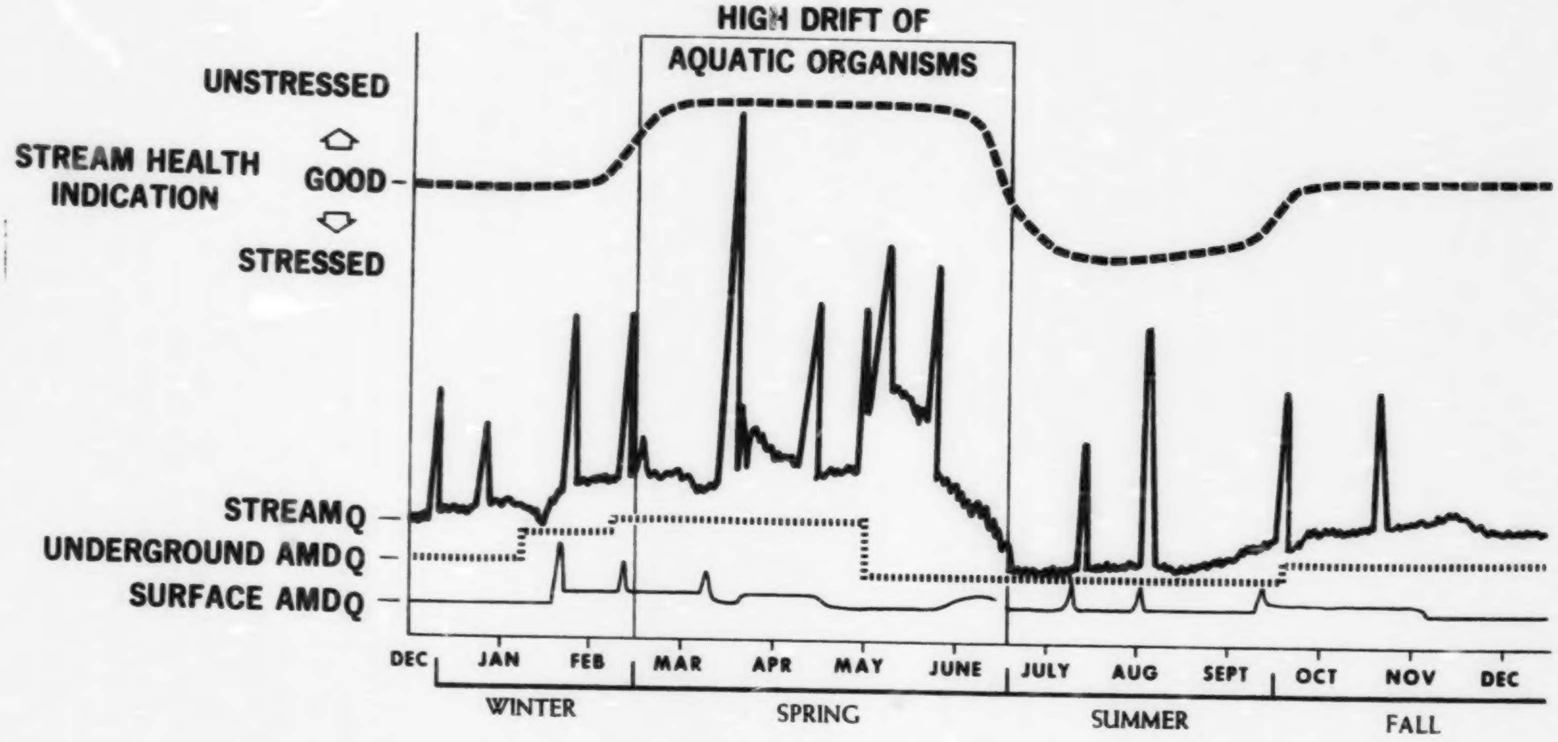
The effect of mine drainage from Champion Run on Indian Creek varied through the study due to seasonal and other influences [Figure 2]. During high discharge periods, the effect of the acid drainage was decreased. This decrease can be explained by the following factors: (1) flow volume from the upstream portion of Indian Creek was at times large enough so that acid drainage in Champion Run was diluted; (2) higher flow volumes from mine drainage sources reduced the concentration of the mine drainage entering Indian Creek, and (3) higher discharges from tributaries, including the upstream portion of Champion Run, further diluted the acid drainage and reduced its influence on the downstream portions of Indian Creek.

During low-discharge periods, the effect of mine drainage from Champion Run was intensified. The acid drainage from the Melcroft mines was only partially diluted by discharge volumes from the upstream portions of Champion Run and Indian Creek. During low discharge periods, the effect of the mine drainage extended further downstream, affecting Station 7. Improvement of water quality occurred below each confluence with an unpolluted tributary. Even though the dilution ability based on stream flow was reduced, the neutralization capacity of the eastern tributaries was improved. Stream flow during low-discharge periods is predominantly from ground water sources. Because ground water in the headwaters of the eastern tributaries flows through calcareous rocks, alkalinity of tributary waters was increased. Thus, the ability to neutralize the acid drainage is also increased.

The relationship between acid mine drainage concentration and stream flow is very complicated and deserves some discussion. Because a significant portion of the mine drainage in Champion Run was from underground sources, these discharges maintained a relatively constant acid mine drainage volume [Figure 2]. Since mine drainage volume is dependent on rainfall infiltration to underground areas, and pyrite oxidation is not appreciably changed by the amount of water present, the

FIGURE 2

Seasonal Conditions of Stream Physical, Chemical, and Biological Parameters



concentration of pyritic oxidation end-products will vary with volume. Because the infiltration rate is greater during the winter, the volume of mine discharges is increased from December through April. Infiltration decreases during the summer months, so mine discharge volumes also decrease.

Additional seasonally related mine drainage discharge problems occur from surface sources [Figure 2]. Pyritic materials on gob piles are well oxidized. During the winter months, reduced surface termperatures reduce the oxidation rates, and temperatures below freezing prevent runoff from gob piles. Initial melt carries the oxidation products into the receiving stream, but high assimilative capacity due to high stream discharge reduces this impact. Chemical reactions on the gob piles are increased during the warm summer months. Rainfall during this period usually occurs as high intensity storms which flush unvegetated areas rapidly. The accumulation of pyritic oxidation end products makes initial runoff highly concentrated, and acid mine drainage slugs precede increased stream flow while assimilative capacity is low. Major damage may occur under these conditions.

Biological Sampling and Analysis

Biological sampling centered on methods to describe the recovery of the stream related both to seasonal differences in stream condition, and to the distance from the AMD source where healthy communities were established. Macrobenthic communities were selected as the best indicators of overall biotic effect from mine drainage. The macrobenthos consists of those aquatic organisms which are commonly found living in or near the stream bottom. These include several orders of insects and a number of other orders of invertebrate animals. Macrobenthos are accurate indicators of present and past stream conditions, because most macrobenthic organisms have long life cycles, and the presence of a certain taxa may indicate conditions over a relatively long period of time. In addition, the sessil nature of most macrobenthic organisms enhances the probability that the effects measured are associated primarily with the area where collections were made.

Samples of macrobenthic organisms from Indian Creek were analyzed in several ways. Details of this analysis can be found in Herricks and Cairns [1974]. One method of biological data analysis will be interpreted in light of the results of hydrologic and water-quality predictions. This biological analysis technique compares species lists from each station, and groups them according to their similarity with other stations. The end-product is a dendrogram showing clusters of stations based on levels of similarity between clusters. This cluster-analysis technique is discussed in detail in Cairns and Kaesler [1969].

The results of cluster analysis are contained in Figures 3 and 4. The dendrogram [Figure 3] contains the cluster of 57 stations collected during May through October, 1971, and Figure 4 contains the cluster of 53 stations collected during March through October, 1972. Analysis of the dendrograms is dependent on selection of a level of similarity which defines the extent of cluster formation. At a limit of similarity of 0.0, one large cluster of all stations sampled is formed, while at a level of similarity of 1.0, all stations are clustered apart from each other. Because the statistical reliability of cluster analysis cannot be rigorously proved, the information in the dendrogram supplies a qualitative description of sample groupings. The level of similarity can be arbitrarily selected, permitting convenient description of the clusters formed in ecological terms. A level of

FIGURE 3

Dendrogram of 1971 Biological Collections

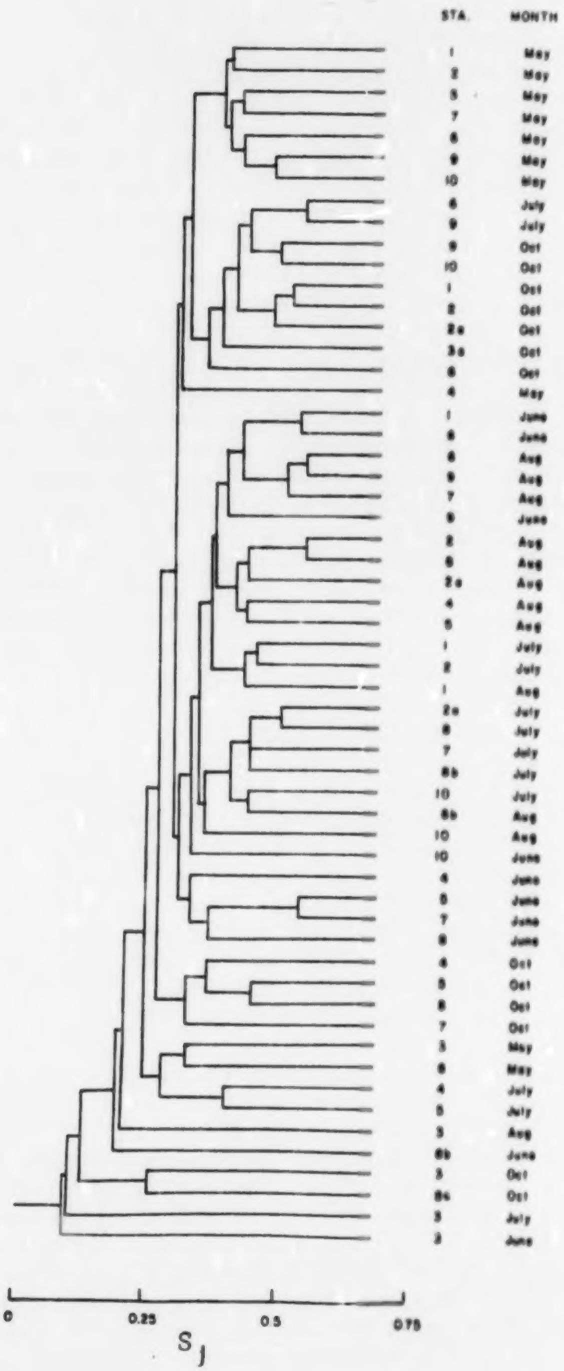
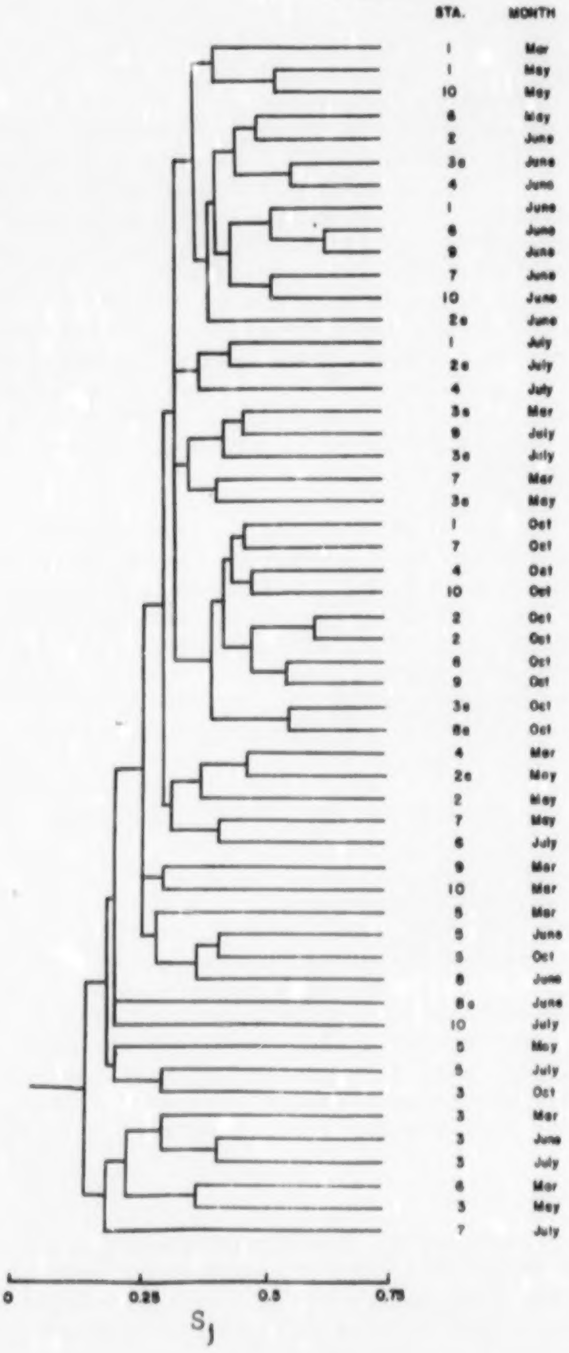


FIGURE 4

Dendrogram of 1972 Biological Collections



similarity of 0.4 was selected. This divided the 1971 dendrogram into five cluster groups of 8, 9, 14, 7, and 19 stations, and the 1972 dendrogram into five cluster grops of 13, 3, 5, 10 and 72 stations.

The major impact of AMD observed in Indian Creek was the chemical alteration of stream water quality and a corresponding detrimental effect on the biota. Dendrograms from both 1971 and 1972 show this impact by clustering stations which were affected by AMD. The AMD tributary, Station 3, and several other stations under direct AMD stress were, for the most part, clustered together. Although Stations 3 and 8a were the only stations located on acid tributaries, Station 5 and occasionally Stations 4, 7, and 8 were included in this cluster due to seasonal differences in AMD effect. The top clusters in the dendrogram contained stations grouped with a high degree of similarity. In general, these included the reference stations (1, 2, 2a), tributary stations (3a, 6, 9), and the recovery station (10). Referring to Figure 2, these results can be integrated with sulfate concentration--indicative of AMD occurrence and intensity--to reveal areas of the stream which are severely impacted. Stations 3 and 5 had high and variable sulfate levels due to AMD input and little dilution. The remaining stations showed decreased AMD influence with lower sulfate concentrations. The variability of sulfate concentrations also was reduced downstream because unpolluted tributaries both diluted and neutralized the AMD.

The dendrograms also can be interpreted in light of seasonal differences related directly to variable hydrologic conditions in the watershed. The first cluster in the 1971 dendrogram is made up of stations sampled in May, a high-discharge period. Although Stations 3 and 8 (AMD tributaries) were not included in the cluster, all remaining stations, both mainstem and tributary, were clustered together. A scan of the dendrogram reveals other cases where monthly samples were grouped together. In general, higher discharge months of May, June, and October were grouped together. The same patterns exist in the 1972 dendrogram, although somewhat disrupted by June flooding.

Another effect of variable discharge, briefly mentioned earlier, relates to variable sulfate concentration due to dilution and neutralization. The mainstem stations below the AMD tributary were clustered with the AMD tributaries during the low discharge periods. Because dilution capacity in the stream system was reduced during these periods, AMD discharges had a greater effect, and demonstrated significant impact greater distances downstream. The seasonal variability in biological impact indicators may not only be related in part to changes in AMD concentration, but also should include other hydrologically related biological phenomena. For example, increased discharge also increases the drift rate of macrobenthic organisms, producing an even distribution of many taxa throughout the stream system. This occurs not only due to increased discharge, but also to improved water quality which makes habitats suitable for recolonization.

Mechanisms of Stream Recovery

The recovery of a stream system from a pollutional stress such as AMD can occur if certain conditions are met. First, conditions must be restored through improvement of water quality. In addition, the seasonal variability inherent in any stream system may at times produce conditions which are suitable for the maintenance of healthy aquatic communities. In many stream systems mechanisms of maintenance and restoration are naturally available. Finally, effecting complete restoration of the stream ecosystem requires a means of restoring aquatic communities. By far the most efficient method of biological system restoration is management of natural processes. For example, studies in Indian Creek indicated restoration of aquatic communities began downstream of unpolluted tributaries. These tributaries provided a source area for organisms which recolonized damaged sections of the stream. Since this recolonization activity was related to stream flow (e.g., downstream drift of organisms), damaged areas can be quickly recolonized if environmental conditions are adequate.

STREAM REHABILITATION

Taking advantage of stream assimilative capacity and the tendency toward natural restoration can be the most efficient means of dealing with non-point AMD discharges. Mining operations, many of which are located in sparsely populated areas, can take advantage of good water quality and avoid the complications of multiple upstream discharges which many other industries face. If care is taken in mine operations, including planning, recreational and other aesthetic benefits of the stream can be maintained while optimal non-degrading use is made of stream assimilative capacity. Planned multiple use then can be a guide rather than a restraint to mining operations for the prevention of major AMD problems [Herricks, et al., in press].

Examples selected for discussion here are the most common types of surface mining operations--steep-slope (contour) mining, and area mining. Each has a different set of problems relating to control of surface and groundwater during and after mining operations, and thus requires different techniques to effect restoration or rehabilitation of stream systems. A final example related to mining activities concerns control techniques applicable to washing, milling, or other mineral benefaction operations producing waste materials which may lead to non-point pollution.

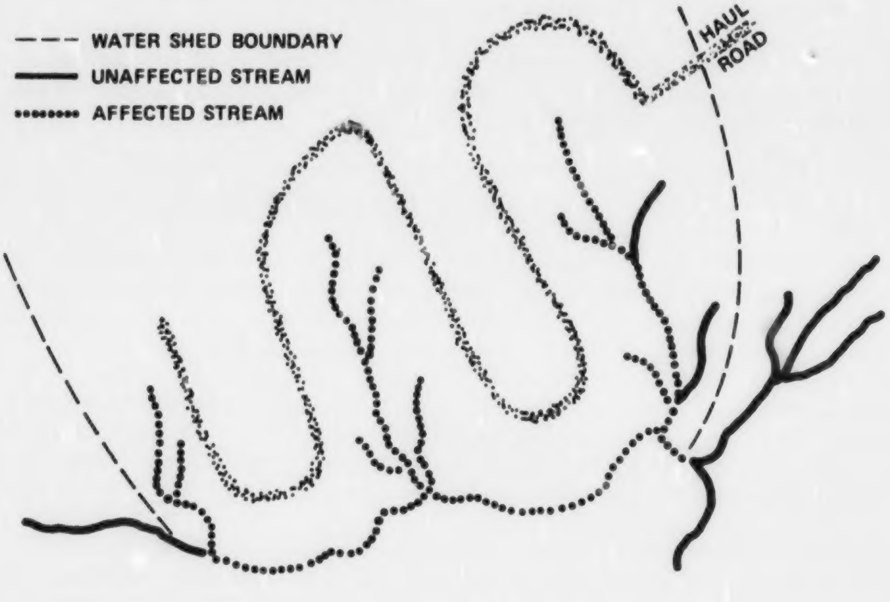
Steep-Slope Mining Operations

In general, mining operations in mountain regions either will follow the seam into the mountain through one or more shafts, or will follow the mineral seam around the surface of the mountain (contour mining). In either instance, major environmental problems exist with disposal of mine wastes or overburden, and control of water in or on the mining operations. Shaft or deep mines often intersect water-bearing strata, and rock fracturing due to blasting--or, in the case of abandoned mines, subsidence--provides routes for surface waters to enter the mine. Where mining operations follow a stratagraphic contour, one or more seams may be mined at different elevations, and each cut may disrupt surface and groundwater supplies.

The control measures associated with steep-slope mining operations are illustrated in part by Figure 5. Key to this control is recognition that watershed boundaries are relatively well defined. Mining in a watershed will, for the most part, severely affect only streams in that watershed. When these streams are overloaded (e.g., assimilative capacity exceeded), potentially damaging materials may be transported out of the watershed downstream to affect a much larger area. The manipulation of terrestrial conditions can fully utilize stream assimilative capacity by minimizing damage and maximizing recovery. Minimizing damage is largely

FIGURE 5

Details of Steep-Slope Mining Operations



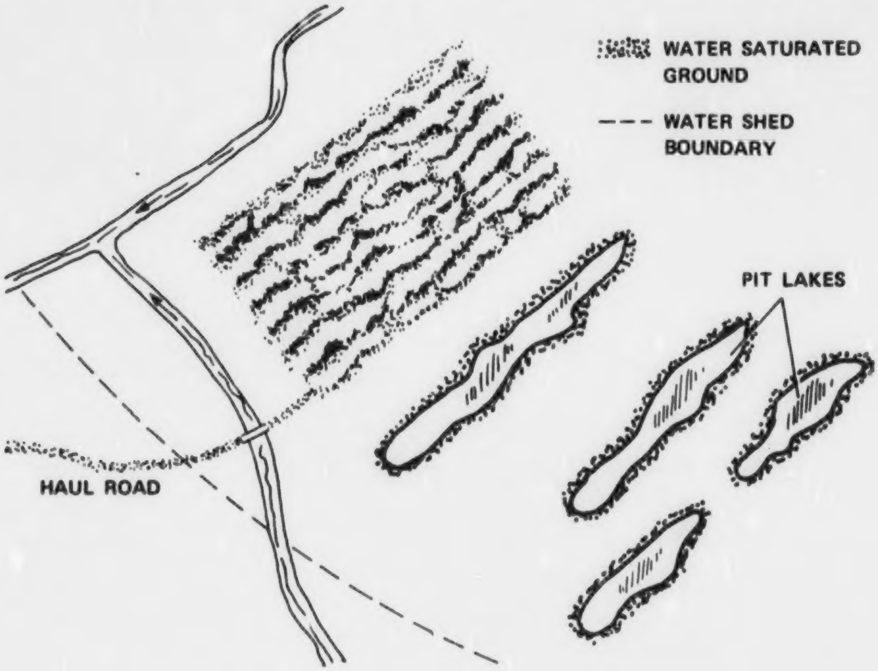
dependent on the conduct of the mining operation and the speed with which disturbed terrestrial areas are reclaimed. Maximizing recovery is more difficult, for it depends on several physical, chemical, and biological interrelationships. Physical-chemical conditions must be improved to provide suitable areas for biological reinvasion and recovery. A straightforward technique for optimizing recovery is maintenance of "biotic preserves," which act as sources of recolonizing organisms. In addition to supplying organisms for recovery, water quality is good in these regions, and flow from these areas may serve to improve deleterious conditions downstream. The location of these "biotic preserves" should be in upstream regions so that natural flow not only improves water quality and downstream habitats, but also provides a path to ease biological reinvasion. By planning mine operations to maintain "biotic preserves," partial recovery of damaged sections of receiving streams can occur rapidly (e.g., 2 weeks to several months). After the stream has recovered, operations in other areas of the watershed can occur using the recovered stream as the new "biotic preserve." Although the efficiency of the total operation may be somewhat reduced, the overall saving in environmental costs may be significan

Area Mining Operations

Area mining operations present significantly different problems than those of steep-slope mining. In steep-slope mining, land disturbance generally is limited to a small percentage of the watershed (e.g., bench and outslope areas), while area mining often disturbs a much larger percentage of the watershed. Although both types of mining have the potential for causing major hydrologic disturbances,

FIGURE 6

Details of Area Mining Operations

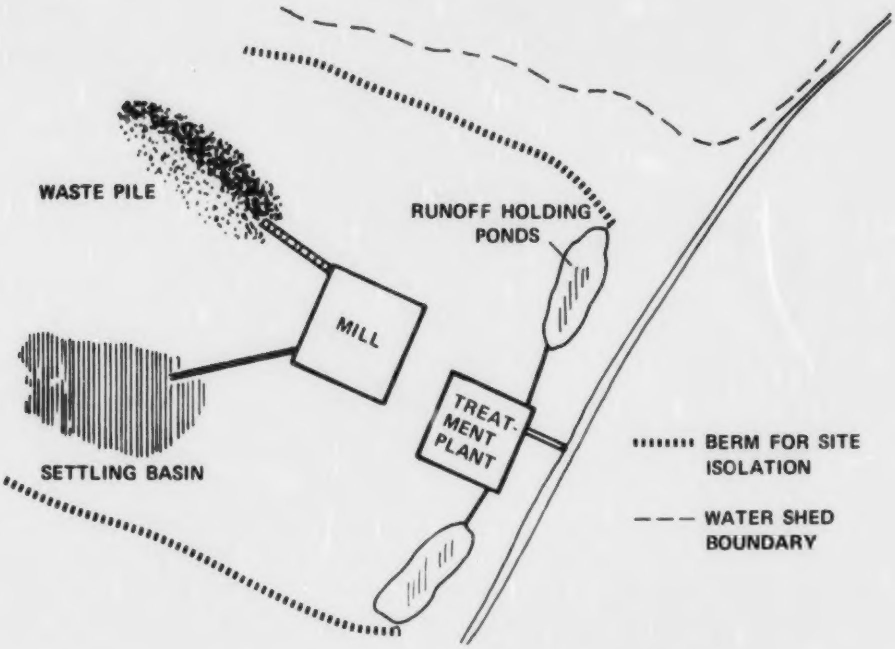


each produces a particular set of problems which must be handled in a different way. Area mining operations have a major effect on watershed topography which may affect hydrology, vegetation, and overall environmental impact. Ore bodies may be completely removed, leaving large pits after mining has ended. Or, in the case of a mineral such as coal (present in one or more seams), overburden materials may be fractured and removed, thus expanding the total volume and resulting in the creation of hills or piles higher than surrounding land. The most destructive feature of area mining is the removal of vegetative cover, which usually leads to increased erosion and the loading of streams with high concentrations of suspended solids. In addition to the vegetative disturbance, the pits created by mining may fill with water, often of very low quality. In the typical situation, one or more margins of the pit lake consist of fractured rock which may have a high permeability. Zones of material saturated with low-quality water can extend outward from the pit lake for a considerable distance, further affecting vegetation. These "reservoirs" of low-quality water may affect both ground and surface water resources over a large area.

A plan view of a typical area mining operation [Figure 6] illustrates some of the problems which must be dealt with in preoperational planning. In general, minimizing ecological damage is the most important part of planning area mining operations; operational procedure is of key importance. Regrading and revegetation should

FIGURE 7

Site Plan for Milling or Washing Operations



closely follow the mining operation. Sediment basins and other control structures should be built to control sediment input to the stream. If toxic water is produced, remedial action (e.g., small neutralization stations) can be taken until regrading or revegetation reduces or eliminates the toxic discharge. Through proper disposal of toxic materials, the water quality of pit lakes can be improved, and compaction and proper grading of disturbed overburden can eliminate surface and groundwater degradation. If damage does occur, recovery can be accelerated by maintenance of "biotic preserves." The preserves or epicenters may be small ponds or large pit lakes with good water quality which contribute to stream flow in an area Fish and other organisms in the lake may move downstream during certain periods, and stream sections with suitable habitats which are supplied with high-quality water will provide the remainder of recolonizing organisms for damaged areas. Special note should be taken of where damage can occur in other watersheds. Haul roads in particular may contribute significant amounts of sediment and may extend across watershed boundaries, causing damage in other streams. Similarly, alteration of groundwater flow by the mining process may reduce surface water flow in other areas, and may even contribute to degraded water quality at substantial distances from the mine. Recognition of these potential problems before mining begins may reduce both long-term costs and environmental damage.

Milling or Washing Operations

The environmental impact of milling or washing operations is as variable as the processes used at each site. Site topography, mill, and disposal area location, as well as discharge variability due to changes in ore composition or milling processes, all affect potential stream damage. In general, damage is caused either by direct runoff from the mill site (non-point sources) or by waste-disposal areas or discharges from one or more of the milling or washing operations (point sources). Tailings, waste piles, or settling basins have characteristics similar to overburden materials (e.g., high permeability), but are potentially more damaging to water resources. These materials often are highly concentrated in tailings or slurry areas. Depending on area geology, construction, maintenance, and operational procedures used in managing these tailings or slurry areas, runoff may have very low quality and significantly degrade receiving streams. Since the mineral usually goes through a series of chemical or solvent treatment steps to extract one or more metals of value, each of these steps produces solid and liquid wastes which may have high toxicity and a high potential for stream damage.

Control measures for milling or washing operations require both consideration of site physical conditions and a detailed analysis of all waste streams associated with the plant (i.e., recognition of both point and non-point discharges). Analysis of site physical conditions must take into account normal runoff patterns and groundwater effects due to movement of large volumes of water through and around the mill plant and settling basins. By artificially creating watershed boundaries to isolate the mill and waste-disposal areas [Figure 7] and constructing runoff holding ponds, surface runoff can be controlled and treated to minimize stream damage. In this way, accidental spills can be contained, and potentially damaging surface runoff can be held for treatment or gradual release. Mill processes should be analyzed and complete flow diagrams developed listing waste-discharge volumes and characteristics. It may be beneficial to mix some waste streams before discharge or treatment (e.g., to neutralize two waste streams, one acid and the other alkaline); in other cases, it may be necessary to isolate certain streams for immediate discharge or treatment. The damage to stream aquatic life can be determined by various bioassay techniques. In this way, waste-treatment operations can be optimized without disrupting the assimilative capacity of the stream.

CONCLUSIONS

The rehabilitation and protection of streams from the effects of acid mine drainage can be effected with proper management of both terrestrial and aquatic resources. From the experimental studies on Indian Creek, it can be established that aquatic communities will recover from stress through natural processes if suitable conditions are provided (e.g., good water quality). When stream rehabilitation from non-point sources of AMD is considered, the process of restoration and recovery can be encouraged through proper management of terrestrial conditions which affect the aquatic environment. For example, control of water on land areas disturbed by mining by means of revegetation or engineering control practices can significantly reduce the impact of the mining operation on the aquatic environment. Similarly, detailed planning of the total mining operation can manage and manipulate the assimilative capacity of a stream to produce the least-cost alternative to waste discharge while maintaining the biological, chemical, and physical integrity of the receiving stream.

Although there is abundant evidence that disturbed natural ecosystems can recover, to at least an approximation of their original condition, it is improbable that they ever will be exactly the same. In some ecosystems, recovery may be slow and and many changes may not be reversable! Amenities often are lost while ecosystems are disrupted. Prevention of damages is far easier and cheaper than restoring ecosystems after damage.

ACKNOWLEDGEMENTS

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SEDIMENT CONTROL DURING MINING AND MINE RECLAMATION

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INTRODUCTION

In excess of 2 million acres of land have been disturbed by surface mining operations in the eastern portion of the United States alone. The result, in many instances, has been excessive soil erosion and severe damage to water resources. The amount of land affected by surface mining operations annually has increased steadily over the years and probably will accelerate even more rapidly in the future as we push towards energy self-sufficiency and are forced to develop lower-grade mineral and aggregate deposits to meet the needs of an industrial and urban society.

Sediment is the major pollutant from surface-mining operations. The effective control of sediment at surface mines is not a simple task. It requires better understanding of control rationale and methodology on the part of the mining industry and sufficient, diligent effort by enforcement agencies.

Extensive research has taken place in this country over the last few decades to advance the state-of-the-art of erosion and sediment control. However, only recently, with the passage of federal and state environmental legislation, has an attempt been made to assemble and digest this wealth of data and develop practical and comprehensive guidelines for planning, implementing, evaluating, and maintaining on-site sediment control practices and procedures.

Considerable work has been performed and continues to be performed in studying and evaluating control practices for surface-mining operations, particularly in the area of mined-land reclamation. However, the major thrust of recent developments in the field of sediment control has been the control of sediment from construction activities and, to a lesser extent, agricultural operations. State-of-the-art studies have been undertaken regarding the control of sediment on construction sites, and numerous pamphlets, reports, handbooks, manuals of practice, and training programs on the subject have been and continue to be developed by federal, state, and local agencies. Much of the methodology for controlling sediment utilized in these fields can be adapted for use during mine reclamation. This paper addresses some of the sediment-control problems arising as a result of surface-mining activity, and discusses methods for controlling sediment during mining and mine reclamation.

TABLE 1

Land Disturbed by Strip and Surface Mining in the United States as of January 1, 1965, by Commodity and State (Acres)

State	Clay	Coal (Bituminous, Lignite and Anthracite)	Stone	Sand and Gravel	Gold	Phosphate Rock	Iron Ore	All Other	Total
Alabama[a]	4,000	50,600	3,900	21,200	100	--	52,600	1,500	133,900
Alaska[b]	--	500	--	2,000	8,600	--	--	--	11,100
Arizona[a]	2,700	--	1,000	7,200	1,200	--	--	20,300	32,400
Arkansas[b]	600	10,100	900	2,600	--	--	100	8,100	22,400
California[b]	2,700	20	8,000	19,900	134,000	--	900	8,500	174,020
Colorado[a]	2,000	2,800	6,200	15,500	17,100	--	25	11,400	55,025
Connecticut[a]	--	--	100	16,100	--	--	--	100	16,300
Delaware[b]	200	--	200	5,200	--	--	100	10	5,710
Florida[a]	13,200	--	25,300	3,900	--	143,600	--	2,800	188,800
Georgia	1,300[c]	300[c]	6,800[c]	1,200[c]	--	--	100[c]	12,000[c]	21,700[a]
Hawaii[b]	--	--	--	--	--	--	--	10	10
Idaho[b]	500	--	700	11,200	21,200	3,100	35	4,200	40,935
Illinois[b]	1,400	127,000	5,700	9,000	--	--	--	--	143,100
Indiana[b]	1,500	95,200	10,200	18,000	--	--	--	400	125,300
Iowa[a]	1,300	11,000	12,200	17,600	--	--	6	2,300	44,406
Kansas	1,100[a]	45,600[b]	7,500[a]	5,100[a]	--	--	--	200[a]	59,500
Kentucky	2,400[a,b]	119,200[a,b]	3,900[a]	1,700[a]	--	--	--	500[a]	127,700

State	Clay	Coal (Bituminous, Lignite and Anthracite)	Stone	Sand and Gravel	Gold	Phosphate Rock	Iron Ore	All Other	Total
Louisiana a/	900	--	100	29,700	--	--	50	--	30,750
Maine a/	400	--	4,400	28,200	12	--	100	1,700	34,812
Maryland	1,200 a,b/	2,200 b/	2,200 a/	18,800 a/	--	--	20 a/	800 a/	25,220
Massachusetts a/	700	--	1,200	36,400	--	--	1,100	900	40,300
Michigan b/	600	--	7,700	25,200	--	--	2,200	1,200	36,900
Minnesota a/	600	--	3,900	41,600	3	--	67,700	1,600	115,403
Mississippi b/	2,700	--	400	26,500	--	--	30	--	29,630
Missouri b/	6,600	31,800	8,400	3,800	--	--	200	8,300	59,100
Montana b/	--	1,500	10	13,500 c/	5,600	100	10	6,200	26,920
Nebraska b/	900	--	4,300	23,700	--	--	--	--	28,900
Nevada a/	100	--	1,600	5,500	5,600	--	600	19,500	32,900
New Hampshire b/	--	--	100	8,000	--	--	--	200	8,300
New Jersey b/	1,400	--	2,000	27,600	--	--	1,000	1,800	33,800
New Mexico b/	13	1,200	100	400	40	--	100	4,600	6,453
New York a/	1,700	--	12,500	42,200	5	--	700	600	57,705
North Carolina a/	5,800	10	6,000	18,400	2,200	300	100	4,000	36,810
North Dakota	800 a/	7,700 b/	300 b/	26,100 a/	--	--	--	2,000 a/	36,900
Ohio	10,200 a/	212,800 b/	21,000 a/	28,100 a/	--	--	4,000 a/	600 a/	276,700
Oklahoma b/	--	23,500	--	2,500 c/	--	--	--	1,400	27,400

TABLE 1 (continued)

State	Clay	Coal (Bituminous, Lignite and Anthracite)	Stone	Sand and Gravel	Gold	Phosphate Rock	Iron Ore	All Other	Total
Oregon[b/]	100	--	300	1,300	6,300	--	10	1,400	9,410
Pennsylvania	10,400[a/]	302,400[b/]	24,400[a/]	23,800[a/]	2[a/]	--	8,800[a/]	400[a/]	370,202
Rhode Island[a/]	--	--	20	3,600	--	--	--	--	3,620
South Carolina[a/]	10,900	--	1,400	10,400	200	8,100	100	1,600	32,700
South Dakota	2,000[b/]	900[b/]	--	28,000[c/]	--	--	--	3,300[b/]	34,200
Tennessee[b/]	2,700	29,300	4,400	18,400	--	27,000	5,300	13,800	100,900
Texas[a/]	6,800	2,900	21,900	122,300	--	--	9,600	2,800	166,300
Utah[b/]	600	--	200	2,200	--	10	500	2,000	5,510
Vermont	--	--	2,300[b/]	4,000[a/]	--	--	--	400[b/]	6,700
Virginia	1,100[a,b/]	29,800[b/]	4,300[a/]	13,100[a/]	600[a/]	100[a/]	7,700[a,b/]	4,100[a,b/]	60,800
Washington[b/]	500	100	1,300	5,700	400	--	20	--	8,820
West Virginia[b/]	300	192,000	2,800	300	--	--	100	--	195,500
Wisconsin[b/]	100	--	9,000	26,400	5	--	49	--	35,554
Wyoming	3,500[a,b/]	1,000[b/]	300[a,b/]	200[a,b/]	--	800[b/]	300[a,b/]	4,300[b/]	10,400
Total	108,513	1,301,430	241,430	823,300	203,167	183,110	164,255	162,620	3,187,825

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a/ Data obtained from Soil Conservation Service, U.S. Department of Agriculture.

b/ Data compiled from reports submitted by the States on U.S. Department of the Interior form 6-1385X.

c/ Estimate.

Source: U.S. Environmental Protection Agency, 1973.

SURFACE MINING ACTIVITY

Surface mining has disturbed over 3 million acres of land in the United States. With increasing demand for coal and other resources, the total area affected by surface mining can be expected to increase rapidly over the next few decades. By the year 2000, it is projected that surface disturbance from all types of mining will increase from 13 million acres to 30 million acres [Environmental Protection Agency, 1973].

The surface mining of coal is the most significant cause of land disturbance from surface-mining activities, as shown in Table 1. Within the eastern portion of the country, such mining activity represents an even larger percentage of the total disturbed acreage. The surface mining of sand and gravel resulted in the second most significant disturbance of land in this region, followed by the mining of phosphate rock, stone, clay, all other mined materials combined, iron ore, and gold [Environmental Protection Agency, 1973].

Several surface-mining techniques are employed in the mining of coal and other materials in the eastern portion of the country. In the Appalachian coal fields, contour mining remains the most commonly used method of surface mining of coal. Today, conventional contour mining is employed only when a stable fill can be constructed on the outslope. Where possible, the fill is carried down to the valley floor to produce a more stable configuration. In addition, some spoil must be retained on the bench for use in reducing the highwall height and sealing the exposed coal seam.

To comply with recent environmental and safety legislation, a number of innovative contour-mining techniques have evolved which have the potential to minimize disturbance to the outslope and also reduce aesthetic damages. These techniques include (1) haulback and blockcut mining, whereby the majority of the spoil is retained on the bench, and (2) head-of-hollow fill or valley fill mining, where some of the spoil, usually the excess spoil resulting from bulking and the preservation of a haulroad along the outcrop, is placed in hollows or valleys on the outslope.

Other surface-mining methods used in extracting coal in the Appalachian region include mountain-top-removal or mountain-top-leveling, and area strip mining. With the upward trend in the price of coal and increasing emphasis on total resource recovery, greater use of the mountain-top-removal method of mining might be expected in the future. This method of surface mining results in the complete removal of the mountain top and the total recovery of all coal resources. While most of the spoil stays on the mountain during mining, some must be placed in head-of-hollow fills to provide working space on the mountain and to minimize overburden handling costs. The overall effect on the terrain is a reduction in local relief and the construction of a rolling plateau suitable, in most instance, for other alternative land uses.

Because of the severe terrain, area stripping is not widely practiced in the Appalachian region. It occurs in some areas of western Ohio and occasionally in areas of Pennsylvania and northern West Virginia. However, in the Eastern Interior Coal Region, including the state of Illinois and the western half of the state of Kentucky, area strip mining is extensively used in the surface mining of coal.

In the mining of anthracite, sand and gravel, phospate rock, clay, and other materials, open-pit mining is generally employed in place of strip mining.

SEDIMENT-CONTROL PROBLEMS ASSOCIATED WITH SURFACE MINING

Coal-mining operations are the largest single source of sediment from surface-mining operations in the eastern portion of the country. This is especially true in the Appalachian region where steep slopes, narrow valleys, highly erodible and sometimes toxic overburden materials, abundant rainfall and runoff, and contour-mining operations all limit the effectiveness of erosion-control and sediment-abatement practices. Although considerable progress has been made in developing new techniques to segregate toxic or highly erodible overburden materials and for vegetatively stabilizing the graded spoils, severe problems still exist in those localities where enforcement efforts are ineffective and in other areas where the operators are not familiar with new control technology.

Returning a contour-mined area to the approximate original slope and drainage pattern may make sense from an aesthetic viewpoint, but such a requirement can sometimes be counterproductive from a sediment-control standpoint. Diversion structures are often necessary to control runoff coming onto and leaving disturbed areas. Where a critical condition exists because of slope conditions or soil properties, the diversion structures should be maintained even after reclamation is complete so that the critical area can become adequately stabilized.

In certain situations, topsoiling is another example of a counterproductive measure. The salvage and proper reuse of good quality, productive topsoil, coupled with prompt seeding and mulching, is an excellent means of quickly establishing a low-maintenance ground cover. However, the use of poor-quality topsoil on surfaces that can be effectively stabilized in its absence, or improper utilization of topsoil, can contribute to increased soil losses and resulting damages from sediment. This partially results from the high erodibility of the fine-textured organic materials found in the soil and the ability of loose organic material to absorb large amounts of water, creating a fluid condition conducive to slippage.

Exposed highwalls containing weathered shales and claystones are another source of sediment from surface coal-mining operations. Severe damage can be avoided, however, by controlling the drainage at the base of the highwall. Where particularly critical conditions exist, drainage control may have to be provided above the highwall as well.

The long, exposed slopes resulting from conventional contour, head-of-hollow fill, and mountain-top-removal mining methods are also a major source of sediment during and following mining operations. The dumping of spoil material in head-of-hollow fills is especially conducive to major water-pollution problems. With this mining operation, the face of the fill remains fully exposed during the life of the filling operation and lengthens as the fill proceeds down the hollow. Additionally, the spoil rests at its angle of repose and may be subject to sliding and sloughage.

The extraction of coal by area strip mining, as occurs in the Midwest and in parts of Appalachia, generally does not result in as much soil loss per acre of land disturbed as other surface-mining methods used to extract coal. Prior to the regrading of the spoil from area stripping operations, the depressions between the ridges or piles of spoil trap much of the sediment resulting from the erosive action of water and wind. During this stage of the operation, the areas most subject to damaging soil loss are found along the perimeter of the mined area. In area strip mining, the reclamation stage of mining is often the most critical in terms of potential soil loss. This is especially true when the disturbed area is to be returned to the approximate original contour and the surface drainage system reestablished. Prompt and effective control measures are required between the completion of grading and the establishment of a vegetative cover if major off-site damage is to be avoided.

Open-pit mining also can produce a significant amount of sediment. The extent of the problem depends on site conditions, the material being mined, the quantity and the nature of waste products resulting from on-site processing operations, and the way the waste material is disposed of. On the basis of the amount of land area affected and the nature of the operations, the open-pit mining of sand and gravel and phosphate rock is especially prone to soil loss and subsequent environmental damage. Where drainage is restricted to the pit area, the problem is less severe, except in those cases where sediment-laden water is pumped from the pit and released, without detention, into the perimeter drainage system. For open-pit mining operations, the major source of sediment is overburden and process wastes disposal outside the pit. The mining and processing of phosphate rock causes significant problems in that the volume of the wastes coming out of the processing operation is about 25 percent greater than the in-place volume of the rock going in.

Soil loss and resulting sedimentation from surface-mining operations near urban areas causes particular problems. Not only is the damage more noticeable, but it compounds the erosion and sedimentation problems resulting from urbanizing activities and usually results in more costly damages than in rural areas. Sand and gravel, stone, and clay mining operations usually are concentrated near major urban centers and, as a result, are of particular concern from the standpoint of sediment control.

SEDIMENT-CONTROL METHODS

While it has been popular to use the term "Erosion and Sediment Control" when describing the technology of controlling sediment from earth-disturbing activities, it is the opinion of some people that "Sediment Control" is more in line with the primary objective of controlling the movement of soil to protect water resources. Certainly there is a concern for protecting land resources as well, especially the offsite damage to vital waterways as a result of increased runoff from disturbed areas, but sediment is the biggest villain. The first and foremost principle of sediment control must be to utilize mining, runoff control, and soil stabilization practices that minimize soil erosion and thus reduce the amount of sediment generated. The second major requirement of effective sediment control is to contain the sediment resulting from unavoidable erosion within the area of disturbance. This requires the utilization of internal control measures to retain the sediment as close to its point of origin as possible, and perimeter measures to provide a "last line of defense" against off-site damage.

Sometimes, there has been a mistaken notion that if internal controls are neglected, the sediment basins employed along the perimeter of the site will provide the required measure of protection. It must be recognized, however, that sediment basins or ponds are not 100 percent effective in removing sediment from runoff. Short of increasing the capacity of the basin or pond, other measures must also be used if additional protection is to be provided. The only fully effective means of controlling sediment is to prevent soil erosion.

There are basically two categories of sediment control technology: erosion control, and sediment containment. The first control category is concerned with controlling sediment at its source, and the second is concerned with preventing sediment from leaving the confines of the mined area.

Erosion Control

Sediment results from erosive processes, either the action of water, wind, or gravity. Controlling erosion on a mining site and thereby reducing the amount of sediment produced is one of the major principles of sediment control. Water and, to a much lesser extent, wind are the major causes of soil erosion during surface mining and reclamation operations. Effectively controlling erosion requires the use of practices that minimize the amount of land area exposed to erosive processes at any one time, the prompt stabilization of exposed soil, and the control of surface runoff which is the principle cause of erosion.

Control of Runoff: This is perhaps the most important means of reducing soil erosion. Along with the impact of falling raindrops, runoff is a major cause of soil erosion. The rate of runoff, including both the quantity and speed of the surface flow, is directly related to the rate of erosion. The physical factors most affecting both the amount and velocity of runoff are infiltration rate, surface roughness, slope steepness, and slope length. These factors can be controlled to a certain extent, thus reducing soil loss. This requires that slopes and drainage systems be properly designed and that both temporary and permanent runoff-control practices be employed during and after mining.

Another means of reducing soil erosion is to intercept runoff before it reaches a highly erodible or bare area, or before it accumulates to a critical concentration, and divert it to a stabilized disposal area. This is accomplished with the use of diversion structures such as ditches, dikes, and terraces.

Surface roughening is one means of reducing the amount of runoff generated during reclamation. Scarification and "tracking" (i.e., movement of a cleated dozer up and down a slope) protect the slope and promote infiltration. Care must be taken to insure that the equipment operator understands that the soil surface must be roughened in a manner that impedes flow rather than channelizing it and speeding its movement downslope.

Surface roughening and the use of diversion structures are principally used to control overland flow. Once the runoff becomes concentrated, grade-control structures may be required to prevent gully erosion or stream-channel degradation. Such structures include downdrains used to channel concentrated runoff down erodible slopes, check dams or weirs which reduce the flow gradient and physically impede channel degradation, channel linings which shield the channel from erosion, energy dissipators to slow the runoff, and level spreaders which spread the flow.

Minimize Area of Exposure: Basically, this requires that the amount of bare soil exposed to erosive processes at any one time be minimized by proper staging or phasing of the mining operations. In this way, perhaps a large area can be mined in stages whereby a number of smaller areas are mined and then reclaimed rather than exposing the larger area to mining all at once, which would expose more bare soil to the elements for a longer period of time.

Soil Stabilization: Another control procedure for use during reclamation is to minimize the period of time that a graded area is exposed by promptly stabilizing the area upon completion of the grading activity. This requires the use of short-term forms of soil stabilization while the long-term forms of stabilization are becoming established. Short-term forms of soil stabilization include the use of chemical emulsions to bind the soil particles into a more erosion-resistant mass, and the utilization of straw, hay, woodchips, and other relatively low-cost mulch materials to shield the soil surface from the action of rain splash and runoff. Fast-growing annual and perennial plant materials which both shield and bind the soil also can be used, while the hearty, perennial plant materials such as grasses, forbs, and shrubs which will be used for permanent stabilization become firmly established.

Sediment Control

To control the sediment resulting from unavoidable erosion, various practices are used to contain or trap the sediment and thus prevent it from leaving the perimeter of the mining area. These practices include the use of small traps to contain a portion of the sediment near its point of origin, filters which remove the coarser sediment fractions, buffers which contain sediment transported by sheet flow, and impoundment structures constructed on the major drainageways to detain runoff and remove additional sediment from the runoff.

Sediment Traps: These are small temporary structures used at various points within the mining area to detain runoff for very short periods of time and trap heavier sediment particles. Examples of such structures include pits dug along ditches and other areas where runoff is concentrated, low gravel dikes placed across graded roadways or in drainage ditches, and straw-bale barriers placed at the base of slopes and in drainage ditches.

Sediment Filters and Buffers: Filtering techniques include preservation of natural vegetative buffers, installation of vegetative buffers downslope from an exposed area, and utilization of gravel or crushed stone filters. The installation or preservation of relatively flat vegetated buffer areas at the base of steep slopes is an effective means of trapping sediment from sheet flow.

Detention Basins or Sediment Ponds: These are the last line of defense in the control of sediment, and are usually constructed on the larger drainageways and near the perimeter of the mined area. These structures are designed to detain the sediment-laden runoff and thus allow much of the sediment to settle out of suspension. Many basins are equipped with perforated riser pipes which allow for the release of the water at a controlled rate. These structures can be very effective in removing suspended sediment if they are properly designed and constructed.

SUMMARY

Present mining operations rely heavily on the use of sediment ponds on the periphery of the mined area to reduce sediment pollution to nearby waterways. Some additional sediment-control techniques adapted from construction industry practice could be utilized during mining. These include: (1) staging of mining for minimum exposure of bare soil; (2) the use of small sediment traps such as straw bales and gravel filter berms in drainage ditches, and (3) use of diversion berms and ditches to keep runoff off exposed soils.

During reclamation, the establishment of a permanent vegetative cover is of prime importance. To this end, erosion and sediment-control techniques should be applied to the reclaimed area. Some of these techniques include: (1) construction of diversion ditches and berms to keep concentrated runoff off the reclaimed area; (2) the use of various types of mulches and fast-growing annual and perennial plantings to protect the area until the permanent vegetative cover takes hold, and (3) surface roughening.

Adequate planning and implementation for sediment control during both mining and reclamation can greatly reduce the amount of suspended solids discharged to the waterways. The use of sediment ponds in conjunction with the other sediment-control techniques mentioned above can accomplish a reduction in pollution from sediment.

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KENTUCKY'S PROGRAM FOR IMPROVED SURFACE-MINING TECHNOLOGY: ADMINISTRATION AND REDUCTION OF ADVERSE ENVIRONMENTAL IMPACT

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INTRODUCTION

Within the past five years, the Commonwealth of Kentucky has embarked upon a research program to comprehensively investigate and recommend new approaches and solutions for dealing with its surface and deep mine-related problems.

When discussing the problems of mining in Kentucky, it is important to note that the state has two very separate and distinct coal fields, the eastern and the western. Both surface and deep mining activities are conducted in each field. The difference in terrain from one field to another is considerable, and while underground mining methods are similar, the types of surface mining practices are comparatively different. On the steeper slope terrain of eastern Kentucky, contour mining is the usual practice, while on the more gently rolling terrain of western Kentucky, coal is surface mined by the method called area stripping.

Contour surface mining commonly is practiced where mineral deposits occur along contours in the mountainous terrain. Basically, this method consists of removing the overburden (soil and rock) above the coal seam by starting at the outcrop and proceeding along the side of the mountain. After the coal seam is exposed and removed by this first cut, additional cuts are often made in a similar fashion until the ratio of overburden to coal brings the feasibility of the mining operation to a halt. This type of surface mining most often creates a shelf or "bench" on the mountainside. On the inside, it is bordered by a highwall, which, in east Kentucky, may range from a few feet to more than one hundred feet in height. On the outer portion of the operation is located the "outslope" which consists of the overburden materials cast down the mountainside during the mining activities. Such outslopes historically have been a constant problem source in eastern Kentucky, and unless controlled or stabilized, the outslopes provide the potential for severe erosion and landslides [U. S. Department of the Interior, 1967].

Area stripping usually is practiced on relatively flat terrain. A trench or "box-cut" is made through the overburden to expose a portion of the coal seam which is then removed. The first cut may be extended, parallel, as far as the limits of the property or the coal deposit. As each succeeding parallel cut is made, the overburden is placed in the area just previously excavated [U. S. Department of the Interior, 1967]. In Kentucky, the surface mine operator is required by law to cover the final mining pit, grade, level, and revegetate the area which has been mined.

At the state government level, there are many problems associated with the management, both physical and administrative, of the mining industry. These problems are discussed below in more detail.

PROBLEMS

Problems associated with the management of the mining industry can be lumped under two separate categories: (1) environment, or those problems concerning the adverse impact to the natural environment as a result of mining, and (2) administrative, or problems dealing with paperwork, communication networks, etc.

Environmental Problems

There are basically four types of environmental problems in Kentucky: (1) sedimentation, (2) slope failure, (3) revegetation failure, and (4) chemical water pollution.

Overall, sediment production is the number one problem associated with surface mining activities in Kentucky. The problem is most acute in eastern Kentucky, but the western Kentucky coal field also experiences this problem.

Slope failure is very noticeable when it happens, often blocking streams and occasionally blocking a road or damaging homes. Since steep outslopes are not generally the practice in western Kentucky, the outslope failure problem is confined mainly to eastern Kentucky.

The problem of revegetation failure is more prevalent in the western Kentucky coal field. These failures are attributed to the following: (1) the generally high sulfur content of the coal; (2) the soil has more general acidic characteristics, and (3) there is a lack of sufficient moisture in the summer and early fall months. Vegetation failures are also found in the eastern Kentucky coalfields. Occasionally toxic materials are encountered, but, more often than not, such failures of vegetation can be attributed to improper reclamation techniques.

Chemical water pollution problems occur mainly in the western Kentucky coalfield, primarily because of those reasons cited for the revegetation failures above. Some chemical water pollution problems do occur in eastern Kentucky, most of which can be attributed to old abandoned underground mine workings and some old unreclaimed orphan surface mined areas (areas mined before 1966 in Kentucky, which was the date of the Commonwealth's first comprehensive surface mining and reclamation law).

Administrative Problems

The major problems in administering the surface mining laws and regulations in Kentucky are as follows:

1. Communications between area field offices and the central office;
2. Data storage and retrieval;
3. Surveillance of surface mine operations;
4. Evaluation of a surface mine operator's performance, and
5. Evaluation of a state surface mine inspector's performance.

RESEARCH AND OTHER EFFORTS

The following discussion centers on various research, demonstration projects and other efforts which Kentucky has used, or is utilizing in its comprehensive approach toward finding answers and solutions to the environmental and administrative problems associated with surface mining in Kentucky.

Reclamation Research

Slope Failure Research: Slope failure occurs as a result of many factors, but mainly is due to the placement of too much overburden material on the natural outslope and to the presence of water. Slides have been known to occur in areas of eastern Kentucky as many as thirteen years after mining operations have ceased. Slope failures are not only eyesores, they drastically increase sediment problems.

Kentucky's Slope Stability Research, funded jointly by the Department for Natural Resources and Environmental Protection and the Appalachian Regional Commission, has been directed toward identifying factors responsible for the residual chronic slides in steep slope terrain, and toward preparation of guidelines for methods of detecting slide movements and practical methods of arresting their progression. Eight representative surface mines throughout eastern Kentucky were selected for instrumentation and monitoring for an 18 month period.

Revegetation Research: The establishment of new vegetation on surface mines is a problem in certain areas of eastern Kentucky. As part of an Appalachian Regional Commission grant program, the Department entered into an agreement with the United States Forest Service Experiment Station in Berea, Kentucky to conduct a study entitled, "Revegetation Research for Eastern Kentucky." Noticeable beneficial results were observed on many of the project test plots compared to the poor-to-mediocore results of the usual revegetative practices performed by the coal operator on problem spoils.

The objectives of the Revegetation Project were as follows:

1. To identify problem areas and classify them by physical and chemical conditions;
2. To carry out soil and rock strata studies;

3. To evaluate present mining and revegetation requirements and recommend alternative means to overcome problems;

4. To conduct tests on research plots and monitor through two full growing seasons, and

5. To develop a manual to be used by surface mine operators and reclamation inspectors or establish guidelines for effective revegetation.

Establishment of quick cover crops has been emphasized. With minimum seed bed preparation combined with addition of fertilizer and mulches, the establishment of quick cover crops such as pearl millet has been successfully demonstrated. However, some problems have been encountered. In some cases, it was found that the quick and abundant growth actually smothered the more desirable perennial grasses, such as Kentucky 31 fescue, which were planted along with the quick cover crops.

The Forest Service group recommended that the Kentucky Division of Reclamation require a chemical and physical evaluation of the overburden strata before issuing a surface mine permit in areas considered highly sensitive to environmental damage, and on coal seams known to have produced acid spoils in the past or suspected of being acid-producers. In cases where the overburden analyses indicate that the proportion of the acid overburden is so great that it cannot feasibly be buried, then consideration should be given to prohibiting mining rather than attempting the very difficult task of correcting the acid problem.

Chemical Water Quality Research (Eastern Kentucky): While chemical stream pollution, particularly acid mine drainage, is not as big a problem in eastern Kentucky as it is in the western Kentucky coal field, some streams in the eastern part of the state nevertheless are affected to some degree by mine drainage pollution, and a few coal seams are known to be acid producers.

Again, in cooperation with the Appalachian Regional Commission, Kentucky began a chemical water pollution project to (1) determine those characteristics which would permit pre-mining identification of areas in eastern Kentucky having a high potential for developing chemical pollution, and (2) to propose alternative waste material handling methods for such areas.

The method employed in this study was to select surface mining sites which were associated with different coal seam groupings, monitor the stream quality, and obtain and analyze stratigraphic samples of coal seams found in each grouping. On the basis of this work, coal seams in eastern Kentucky that have the highest potential for acid production have been identified, and appropriate methods of overburden handling have been outlined.

Chemical Water Quality Research (Western Kentucky): Many of the area-type surface type mines in western Kentucky operate with the pit floor well below the local water table. Consequently, both ground and surface water accumulates in the pit where the chemical reactions leading to acid formation take place. In the past, drainage water which accumulated in the surface mine pit was discharged untreated directly into a stream. However, since the adoption of Kentucky Water Quality Standard for Surface Mining, this method of discharging water is no longer permitted. Water which has accumulated in a pit must now meet stringent standards before being discharged. Concurrent with an oversupply of acid water, revegetation efforts sometimes fail in western Kentucky during some seasons of the year because of insufficient moisture in the soil.

In order to attack both of these problems concurrently, Kentucky, in cooperation with the Environmental Protection Agency and the Peabody Coal Company, began a project entitled, "Revegetation Augmentation by Reuse of Treated Surface Mine Drainage." The objective of this ongoing project is to demonstrate the technique of using neutralized acid mine drainage from a full scale, active surface mine to irrigate new vegetative cover on regraded spoil banks, and to eliminate pollution from the surface water.

Both the neutralization and irrigation units will be engineered and constructed for semi-portability, so as to be compatible with the transient nature of the surface mine operation. Continuous monitoring of discharges, receiving streams, and runoff quality will be carried out during the entire project. Costs and effectiveness of the project will also be documented.

Demonstration of Debris Basins for Control of Sedimentation in Steep Slope Terrain: Under present regulations, the surface mine operator in Kentucky is required to build approved sediment control structures in the hollows below the mining site. These control structures are designed to retain the water coming off the strip mined area long enough for the suspended solids to settle out. While debris basins are the best available practical means of controlling sediment from strip mining, there are still many limitations and problems with this method of sediment control.

Under a grant from the Environmental Protection Agency, Kentucky has undertaken the task of developing the needed research and documentation on the current practices of using debris basins for sediment control. In this study, Kentucky will evaluate and document data relating to the design, construction, operation, maintenance, costs, and the quality of discharge water coming from control structures presently used in eastern Kentucky. Subsequently, based on data obtained from this study, alternative design and construction methods will be developed applicable to both producing the desired water quality and insuring the safety of people living downstream.

Demonstration of On-Site Control of Surface Mine Sedimentation: In view of the problems and limitations of controlling sediment by the use of debris basins, the Environmental Protection Agency has approved a grant for a project which will demonstrate the cost and effectiveness of using non-structural conservation practices at the source of the sediment in connection with the modified block cut method of contour mining. These conservation practices will present no flood hazards, and hopefully will prevent the sediment from reaching the stream channel, thus attacking the problem at its source, not after it is already in the stream.

Water diversion terraces will be located above and below the disturbed area. These diversion terraces will be constructed and vegetated prior to the beginning of any mining activity. The outlets of the diversion terraces will be broad, vegetated "fan" areas. The "fan" areas will also be constructed and vegetated prior to mining, and will serve as the final filtering device before the water reaches the stream channel.

After the vegetation is established in the diversion terraces and "fan" areas, haul roads will be built to each area prior to the removal of the overburden.

After the coal and overburden from the first mined area is removed, the overburden from the second block to be mined will be placed in the vacated area, returning the ground to its approximate original contour. The spoil will then be mulched and seeded immediately. The slope length of the replaced spoil will be reduced by specially designed terraces.

This method, if demonstrated satisfactorily, will not only minimize the disturbed area and reduce the volume of sediment leaving the operation, but will also eliminate the highwall and the outslope--thereby returning the area to nearly its original state and reducing the potential for slides.

Demonstration of Coal Mine Haul Road Sediment Control Techniques: The design, construction, and maintenance of haul roads have not been of very much concern to many coal mine operators since their main interest is in the extraction of the mineral itself. Consequently, many haul roads have been poorly designed and grossly neglected. However, in terms of the environmental impact, the haul road is a principal contributor to stream pollution. Some studies indicate that as much as half of the sediment coming from a surface mining operation can be attributed to the haul road.

Although general guidelines concerning haul roads have been issued by various regulatory authorities, they have not been definitive, nor have any innovative techniques been introduced. In order for Kentucky to better provide definitive criteria for the control of sediment from the haul roads, the state has begun, along with the Environmental Protection Agency, a project entitled, "Demonstration of Coal Mine Haul Road Sediment Control Techniques." The haul road under study will be divided into various test sections which will be approximately one-fourth mile in length, and separated by natural drainage divides. Various control techniques to be introduced into the test sections include such things as diversion ditches above the road's highwall, slope reduction of the highwall, low-cost drainage control structures, grassed or lined ditches, and soil cements. The results of these control methods will be monitored for each section by on-line monitoring equipment and periodic grab samples.

Surface Mine Pollution Abatement and Land Use Impact Investigation: The reclamation research effort has concentrated attention on specific problems, limited areas or, at most, an entire mine. In contrast, a new program sponsored by the Kentucky Department for Natural Resources and Environmental Protection and the Appalachian Regional Commission is entitled, "The Surface Mine Pollution Abatement and Land Use Impact Investigation." This program is aimed toward three main goals: (1) identifying pollution problems resulting from surface mining on a large watershed, (2) recommending alternative abatement schemes, and (3) analyzing the impact of surface mining on surrounding land use patterns. The investigation will be carried out on a selected model area in the eastern Kentucky coal field. It is anticipated that knowledge obtained from this project will be applicable throughout the Central Appalachian Region.

During the project an inventory of orphan (pre-law), active, and inactive surface mines will be undertaken. Both high and medium altitude aerial photographs will be employed as the primary inventory tool, with periodic spot checks on the ground. This inventory will provide information leading to the identification of land and water problems associated with surface mining and enable a model area to be selected for more thorough investigation.

After the selection of the model area (probably a manageable subtributary of a water basin), problems will be classified by type, degree, and extent. Surface mining impacts upon existing land use and population centers will be evaluated in terms of environmental, social, and economic values.

Within the model area, the potential impact of surface mining upon future land use, especially in areas of potential growth, will be projected and evaluated, and potential areas of critical environmental concern will be identified. Alternative abatement programs and their cost will be determined for those problems determined to be significant.

Design of Surface Mining Systems: Because of the obvious problems involved in continuing and possibly expanding the surface mining industry, it was felt that Kentucky needed to identify new approaches toward surface mining, and at the same time take a look at present methods by which the Department for Natural Resources and Environmental Protection administered the surface mine law and regulations. Because of this need, a study was undertaken entitled, "Design of Surface Mining Systems," in conjunction with MATHEMATICA, Inc., and the engineering company Ford, Bacon and Davis. The study was divided into two parts. Part "A" of the project was aimed at identifying alternative approaches to surface mining, using varying combinations of existing equipment which appear to be feasible, based on different resource objectives such as maximizing recovery of the resources, minimizing the adverse environmental impact, and reclaiming land for multi-purpose use on a large scale. In addition, the contractor was asked to recommend desirable changes in the general administration of Kentucky's reclamation enforcement program. Part "B" of the study was directed (1) toward identifying feasible modification of mining systems which might improve surface performance, and (2) toward the design of new equipment, procedures, and systems which would improve the mining operation, both for increasing resource recovery and lessening the adverse environmental impact.

From the investigations conducted under this program, two new modified mining methods were proposed. One is the modified block-cut method, which is presently proposed for implementation through an Environmental Protection Agency demonstration grant.

Another potential method which appears feasible is a modified mountain-top-removal mining technique. This mining operation is conducted in a similar fashion to the conventional mountain-top-removal method, but, like the buried highwall technique, it substantially reduces the need to deposit overburden material on the outslopes typical of mining in Eastern Kentucky.

Other Efforts

Coal Utilization Research: The 1972 Kentucky General Assembly appropriated $400,000 to support coal utilization research at the University of Kentucky's

Institute for Mining and Minerals Research. During the first two years of the program, main emphasis was on (1) the availability and detailed characteristics of Kentucky coal with respect to process evaluation and site selection for coal liquefaction and gasification plants, and on (2) the establishement of a nucleus of people with expertise in the broad area of coal utilization.

In 1974, the General Assembly approved an appropriation of $57.7 million to mount a major expansion of Kentucky's energy resource utilization program. The appropriation provides: $50 million over a six-year period to provide seed money for pilot and demonstration plants to produce synthetic fuels from coal; $4 million for a new specialized research and analysis facility near Lexington in central Kentucky, and $3.7 million for expansion of the research efforts through the University of Kentucky's Institute for Mining and Minerals Research.

The program includes projects in the five following areas: evaluation of reserves; mining; reclamation; coal utilization, and environmental effects. All five areas of research are being addressed in terms of short-range practical applications to immediate problems and with a view toward longer-range developments.

All of these projects have a direct bearing on environmental quality, especially the work on coal utilization. While some of the interest in synthetic fuels from coal stems from the convenience of using liquid or gaseous fuels, it should be kept in mind that the removal or abatement of impurities, principally ash and sulfur, is a major underlying factor motivating much of the current research efforts to develop clean fuels from coal.

Reclamation of Orphan Mined Land: Nearly 130,000 acreas of land were strip mined in Kentucky for their coal deposits prior to 1966, and most of this area was abandoned before the first effective reclamation laws were passed in 1966. No accurate data are available for the state as a whole, but estimates of the amount of inadequately revegetated land remaining range from 70,000 acres to 93,000 acres. The latter figure includes other surface mining activities such as stone, clay, and sand and gravel in addition to coal. Much, if not most, of the revegetated land has been reclaimed by nature. The usual criterion is to consider a vegetation cover of 70 percent or more to be adequate.

In 1972, the Kentucky General Assembly established a revolving fund to reclaim orphaned strip mine land. The Legislature appropriated $500,000 to implement the effort during the following two years, and added $1,000,000 in 1974 to continue the effort. The Kentucky Department for Natural Resources and Environmental Protection administers the orphaned mined lands rehabilitation program.

The goal is to purchase the land and rehabilitate it as rapidly as possible. Two areas have been obtained, both in the western Kentucky coal field. One area comprising about 225 acres in Muhlenberg County has been designated as a Federal Job Corps Training Camp where approximately 100 men are receiving on-the-job training as heavy equipment operators while they are reclaiming the land. This cooperative effort between the U. S. Department of Labor and the Kentucky Department for Natural Resources and Environmental Protection has progressed to the point that dormitories, maintenance shops, and other buildings have been finished, and reclamation of the first 75-acre block should be completed by June, 1975. The reclamation plan envisions about 15 acres being set aside for the Job Corps

Training Center and the remainder to become permanent pasture.

The 400 acres of the Hopkins County site lies along one side of a State highway connecting two growing communities. One, the county seat, has a population of about 16,000, and the other 25,000. Because of its location and accessibility, it is planned that the restored land will be used for residential, industrial, and recreational purposes.

Land restored under this program can be transferred to other state agencies, sold at cost of acquisition and restoration to political subdivisions of the state, or publicly sold to the highest bidder. But, in any case, sale of the land is under the condition that no strip mining can be carried out on the land.

Legislation: As a result of Kentucky's efforts, two major pieces of mining legislation were passed by the 1974 Kentucky General Assembly:

1. New regulations authorized by the 1974 Legislature to be effective in July, 1975 will permit no more than 40 percent of the first cut spoil to be pushed over the outslope, and no overburden due to second and subsequent cuts can be placed on the outslope or on the outer one-third of the bench. At present, slopes in excess of 27 degrees can be mined only by augering or by a haulback method where no overburden is allowed over the outslope.

2. The 1974 Legislature also directed the Department for Natural Resources and Environmental Protection to regulate the surface effects of underground mines. Regulations are now in the development stages requiring permits and reclamation of haul roads, cuts for entry, parking areas, waste disposal areas, and other surface disturbance associated with underground mines.

REFERENCES

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Control Remedies, Policies, Programs

NON-POINT WATER POLLUTION:

FEDERAL AND STATE PERSPECTIVES

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INTRODUCTION

I appreciate the opportunity to discuss with you where we are and where we are going, as federal and state governments, in managing our land resources to reduce and minimize pollution from runoff. I have been asked to discuss "Enforcement Programs and Attitudes on Non-Point Source Pollution." I would rather address the subject of management of our land and water resources to meet water-quality goals, in balance with the renewable and non-renewable resource production goals of our national economy.

POINT AND NON-POINT SOURCES

Before I proceed, we should define our terms. The Federal Water Pollution Control Act defines "point source" as any "discernible, confined and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal-feeding operation, or vessel or other floating craft, from which pollutants are or may be discharged." It is, and was meant to be, a fairly inclusive definition.

"Non-point sources" were not defined in that statute, except by reference that they were related to activities specified in Sections 208 and 304. The legislative history in the Senate contemplates the Environmental Protection Agency (EPA) developing a fairly prescribed definition of non-point sources. To date, we have not done so. I think, however, in this conference, we might be able to develop some useful criteria without any reference to statutory definitions. I suggest we start with the following:

1. Non-point sources enter the receiving waters in a diffuse manner.
2. They are intermittent.
3. The pollutants arise over an extensive area of land and are in transit over land before they enter navigable waters.
4. Non-point source pollutants generally cannot be monitored readily at their point of origin and are not always traceable to their exact source.

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5. Their prevention or control must be directed at a site-specific management or conservation practice.
6. Compliance monitoring for non-point sources is conducted on land rather than on water.
7. Non-point sources cannot be measured in terms of effluent limitations.
8. The extent of non-point source pollution relates, at least in part, to certain uncontrollable climate events.
9. From an institutional management point of view, man's activities which cause non-point pollution are typically non-repetitive processes on extensive land areas, as contrasted with repetitive operations on smaller but intensively used land areas.

LEGISLATIVE HISTORY

A 21st-century historian observing the fate of the environmental movement of the last third of the 20th century will, I predict, note that the Federal Water Pollution Control Act Amendments of 1972 were a cataclysmic episode in our efforts to rehabilitate the nation's waters. Of most importance, the Act supplied to federal and state governments effective tools to regulate the discharge of pollutants from municipal, industrial, agricultural, and mining point sources. It required the issuance by EPA of national effluent guidelines. It required each discharger to apply for a permit to be issued by EPA or by those states which applied for the permit program. Effective enforcement authority by state and federal government, and by citizens themselves, was clearly chartered. The 1972 Amendments clearly establish the priority for the federal and state governments to control point-source discharges.

To date, EPA and state water-pollution control agencies have been governed by this priority and have allocated almost all of the water-pollution control resources to the achievement of point-source clean-up goals. In the 42 months since passage of the Act, EPA and the states have issued 32,383 permits to dischargers. We have issued 16,413 industrial permits, 12,895 municipal permits, 1,739 agricultural permits, and 1,126 federal facility permits. As of March 31, 1975, we are over 90 percent along the road to issuing "major" permits to municipal and industrial dischargers. This is truly a creditable achievement by both federal and state agencies.

The 1972 amendments also laid the first federal statutory base for addressing non-point problems. The Senate Committee on Public Works wrote legislative history for our national non-point source pollution control program in their 1971 committee report on S2770 [S.R. No. 92-414, 92nd Congress, 1st Session, 39]:

> "It has become clearly established that the waters of the nation cannot be restored and the quality maintained unless the very complex and difficult problem of non-point sources is addressed... for the first time the committee bill provides a mechanism to establish a program to control the principal non-point sources of water pollution. Section 304 requires that the Administrator develop, in conjunction with the other appropriate federal agencies, information regarding non-point sources and their control.

> Following the publication of this information, each state or regional planning agency under Section 209 (208) is required to develop 209 (208) plans for non-point source pollution control after public hearings, and submit that plan, by a certain time to the Administrator. Specific non-point sources must be identified within such a plan and then in each instance the state must identify each of the specific sources within each category. The Committee recognizes, at the outset, that many non-point sources of pollution are beyond present technology of control. However, there are many programs that can be applied to each of the categories in non-point sources and the Committee expects that these controls will be applied as soon as possible.

EPA's role in non-point source control management is to develop overall strategies, to involve other federal agencies, to suggest technical and institutional approaches to states, to develop pilot projects of both technical and institutional concerns, and to establish regulations and guidelines to the states to achieve the 1983 goals.

THE STATES' ROLE

Section 208 of the 1972 Amendments is the key for state program development for the control of non-point sources. Under its provisions, areawide and/or statewide water-quality management plans must be developed and implemented within each state. State or local planning agencies are designated, usually by the governor, to develop and implement a "208 water-quality management plan." The policy established by Congress is that the control of non-point pollution is a state and local responsibility.

One primary thrust of the 208 program is to provide for areawide waste treatment for urban, industrial, and other areas having substantial water-quality control problems. The other primary thrust is to provide for a process to identify and control non-point sources from agriculture, forestry, mining, construction, salt-water intrusion, disposal of pollutants in subsurface excavations, and all well activities. We expect approximately 150 areas will be designated and funded by June 30, 1975, with 100 percent grants to state or local agencies ranging from $200,000 to $10 million for the two-year planning period. These areas will address pertinent forms of non-point sources in the development of management plans. The Act also provides for the state to plan and manage all parts of the state not designated under areawide management. In addition, for the management of non-point source pollution, the governor has the option of planning and managing non-point source controls on a statewide basis.

The development of non-point source plans by the state under Section 208 requires assessing the state's non-point source problems, establishing priorities for non-point source categories, determining best management practices for priority problem areas, developing long-range scope of activities for lesser priorities (designed to meet 1983 water-quality goals), and establishing institutional mechanisms for achieving prevention and abatement requirements. These plans are to provide the decision basis for the action program and will include the identification or evaluation of elements such as:

- Type of non-point source problems
- Waters affected
- Seriousness of the effects of those waters
- Non-point sources contributing to the problem
- Measures necessary to produce the desired level of control through application of "best management practices" (or more stringent control measures as appropriate)
- Period of time, cost, and proposed regulatory programs necessary to achieve the desired controls
- Implementing and operating agencies needed to achieve the controls.

FEDERAL AGENCIES

The federal establishment, outside of EPA, has two statutory responsibilities to assist the nation in achieving the non-point source, as well as the point source, water-quality goals of the Act. First, federal agencies having property or facilities engaged in any activities which result in the discharge or runoff of pollutants shall comply with appropriate federal, state, interstate, and local requirements for control and abatement of pollution. Therefore, we expect that federal agencies, to the extent that their budgets permit, will take the leadership in the management of their own land to reduce pollution from non-point sources. We also expect federal permitting and grant-assistance programs to assure that their clients are in compliance with appropriate EPA regulatory requirements.

Second, under Section 304(j) of the 1972 Act, EPA has entered into agreements with the Secretaries of Agriculture, Army and Interior to provide support for these agencies' programs to develop and implement plans under Section 208 of the Act.

We have begun the federal interagency program development process. Our basic vehicle is Executive Order 11752. We have begun to sort out the critical program relationships with several agencies.

In approaching solutions to agriculture and forestry-caused non-point sources, we are indeed fortunate, as EPA Administrator Russell Train noted recently, that water-pollution control and soil and water conservation require identical measures in most instances. The control of soil erosion, for example, though it cannot prevent the loss of some soluble compounds, does keep waterways free of sediments with their attached, insoluble pesticides and nutrients. And pollution control itself is a vital part of any water-conservation project.

To this end, the Environmental Protection Agency encourages the agricultural community to implement conservation practices, universally and consistently, as recommended by the Soil Conservation Service for the past 40 years. This alone would be an enormous step toward a healthier environment and higher farm productivity in the decades ahead. In order to achieve reduction of sediment and associated pollutants, a greatly expanded cooperative effort by environmental and agricultural agencies at all levels is being initiated.

With the Soil Conservation Service, EPA is developing program-coordination efforts which include potential utilization of the SCS field-manual system as a part of the federal/state sediment and agricultural pollution-control program, the involvement of SCS technical assistance and educational delivery systems in the 208 planning and management process, the integration where feasible of regional water-quality management and resource conservation and development programs, and joint appraisal of the need to control sediment from rural roads.

With the Forest Service, EPA is evaluating the timber sales contract process to see to what extent improvements can be effected in water-quality management methods, forest practices legislation, and expanding Forest Service technical assistance to states in the water-quality area. The process includes reviewing both agencies' planning processes to see where watershed, land use, and timber management planning of the Forest Service can interrelate with state and areawide 208 water-quality planning, and developing a cooperative water-quality monitoring program.

We also are developing communications and working relationships with the Extension Service, specifically the Extension Committee on Organization and Policy, with which we hope to continue to emphasize water quality as an integral part of the technical education programs offered by the nation's extension services. These programs already have played a key role in introducing appropriate pollution-control measures for animal feedlots. Extension services throughout the country are carrying out pesticide training and certification under the 1972 Federal Insecticide, Fungicide and Rodenticide Act.

We recently concluded an interagency agreement with the Department of Housing and Urban Development to provide a framework for state and local land-use planning coordination with the water-quality planning effort under Section 208.

With the Coastal Zone Management program of the National Oceanic and Atmospheric Administration (NOAA), we currently are engaged in the appropriate development of interagency program guidance.

With the Departments of Interior and Agriculture, we soon will be engaged in cooperative development of strip-mining regulations and integration of state strip-mine control programs with 208 program management and other requirements of the 1972 Act.

This is only a part of the interagency cooperative program being developed by EPA to achieve the necessary reduction of non-point source pollution to achieve the nation's water-quality goals.

TECHNICAL AND INSTITUTIONAL GUIDANCE

EPA is required to issue technical and institutional guidance as a continuing process. The first generation of technical guidance was issued last October and includes eight publications devoted to describing the processes, procedures, and methods for assessing and managing pollution resulting from the activities described above.

We have also developed, in conjunction with other agencies, suggested state implementation legislation, such as the suggested sediment control act and are now proposing a suggested forest practices act. We have under review now second-generation technical guidance currently being developed by the Agricultural Research Service of the U.S. Department of Agriculture and by the Midwest Research Institute for assessing and controlling non-point sources. Other guidance documents are being developed in our regional offices as a part of a pilot program. We are exploring the utilization of technical manual systems of other agencies such as the Soil Conservation Service and Extension Service--manuals which would have built-in appropriate water-quality management guidance.

MANAGEMENT AND ENFORCEMENT

Under the 1972 Act, the states have full responsibility for management and enforcement of non-point source programs. EPA has no reserve or step-in authority or any realistic punitive mechanism to force a state to establish an appropriate program.

Several states already have established mandatory non-point source regulatory programs. One report available at this conference lists state sediment-control institute programs issued by EPA in April, 1975, and summarizes the efforts of the National Association of Conservation Districts, EPA, and the Soil Conservation Service to hold institutes in 40 states to discuss regulatory and mandatory actions for control of erosion and sediment. The basis of these institutes was a suggested act for soil erosion and sediment control developed in 1972 under the auspices of the Council of State Governments by a committee chaired by the Speaker of the Iowa House of Representatives. These institutes resulted in the drafting, redrafting, introduction and, in several cases, passage of legislation establishing regulatory programs. When the institutes began, five states already had acted. To date, 12 states have taken one or another type of legislative or executive action. New legislation has been prepared in 15 other states. Efforts to enact previous bills were accelerated. Let me cite three examples.

Iowa was the first state to pass sediment-control regulatory requirements that affected agriculture. The basis of the regulation is a required soil and water conservation plan. Enforcement provisions are based upon actual soil movement and damage to either a waterway or adjoining property.

Under its water-quality statute, Pennsylvania adopted a strong, mandatory sediment-control program which covers all earth-disturbing activity. Construction now is subject to the requirements of an on-site conservation plan drafted by a technically competent person. Agriculture will be subject to the regulations in January, 1977. There is a size threshold level of applicability, and a distinction is made in the regulation between soft and hard enforcement. The soil conservation districts, together with the state soil and water commission, have the administrative and soft enforcement responsibility. So-called hard enforcement is primarily the responsibility of the water-quality agency. However, conservation districts have the option of requesting the hard enforcement responsibility as well, and over a quarter of the soil conservation districts in Pennsylvania have so requested. There have been several enforcement actions under this program.

Ohio has a similar program, but its sediment-control law places the entire responsibility on the shoulders of the conservation commission and the soil conservation districts. The governor has requested a $4 million budget for the administration of this program in the current session of the legislature--half for administration and half for cost-sharing for necessary practices.

Maryland and Virginia have statutes and regulatory requirements which apply primarily to construction activities. Observers of the Maryland program have estimated that the regulations are about 30 percent effective. The weaknesses are in administration, inspection, and enforcement.

EPA, with the Forest Service, is developing a similar program as one option for states to consider in managing pollution from forest lands. During the past few months, we have been developing a suggested forest practices act in conjunction with the Council of State Governments. The regulatory basis of this act is a mandatory on-site plan, or, in the case of small landowners, a notification-of-activity requirement. This suggested act has aroused considerable concern in the forest industry and land ownership communities. To develop an understanding of program need, EPA, together with the Forest Service, has requested that the American Forestry Association hold regional institutes similar to the sediment-control institutes.

Washington and California already have implemented a forest practices act requiring a permit, and Maine is considering one this year.

To control pollution from mining, several states have issued very strong mining statutes and regulations. The Surface Mining Control and Reclamation Act of 1975, as it was reported out of the House, lists effluent guidelines for pollution from active mines. EPA would be required to work closely with the Department of Interior to develop these regulations. In addition, the Act calls for a 30 cents-a-ton tax on newly mined coal to establish a reclamation fund to minimize water pollution from abandoned mines. EPA and the state pollution control agencies would work closely with the Soil Conservation Service in administering the reclamation program.

Most 208 agencies are beginning to focus on urban runoff, and I look to Paul Barker of the New Castle, Delaware agency, to discuss this in his presentation at this conference.

TIMING FOR THE NON-POINT SOURCE MANAGEMENT PROCESS

Our about-to-be-issued third edition of National Water Quality Strategy sets out our objective for the sequence in the development of the programs by the states. This sequence of timing may take one of two forms, depending on whether or not areas are designated by the states as 208 areas.

Timing in areas not designated as 208 areas (and designated areas for which the state has retained non-point source planning and management responsibilities) is as follows:

- In 1975, the state and the EPA Regional Administratior will agree on a level of detail and the timing for preparation of State Water Quality Management Plans.

- As planning proceeds, states will refine their non-point source problem assessments which they addressed in their first Section 305(b) report, and define their non-point source control needs and regulatory programs.
- Beginning in April, 1976, more specific timing of state implementing actions identified in State Water Quality Management Plans will be reflected in Annual State Strategies.
- Specific controls and regulatory programs will be developed by the states on a priority basis as soon as adequate information, resources, and legal authority become available.
- All regulatory actions will be initiated in time to achieve the 1983 water quality-goals.

Timing in designated areas--where the state has not retained non-point source planning and management responsibilities--is simply stated:

- In designated areas, program planning is to be completed in the first two years after designation. Implementation of regulatory programs is to be completed in time to achieve the 1983 water-quality goals.

CONCLUSION

I would like to condition what I have said above for several reasons. First, a recent court decision concerning the feedlots upsets our administrative definitions of point sources for urban runoff, irrigation, agriculture, feedlots, and silviculture [Natural Resources Defense Council v. Train, U.S. District Court for Washington, D.C., Civil Action No. 1629-73, March 1975]. We now are trying to regroup, and are seriously evaluating the remaining options open to us under this decision. The other qualifying aspect is that Congress built two assessment provisions into the Act. Congress established the National Commission on Water Quality to evaluate EPA and the states' performance in the first three years of the Act, making recommendations on the feasibility of the programs under the statute in reaching the 1983 and 1985 goals. Most important, the Congress built into the Act an annual reporting requirement by the states to assess non-point sources and to recommend possible solutions. Another limiting condition is that the Congress provided no additional funds for this program-management activity by the states, nor were funds provided for grants or cost sharing for the installation of best management practices. The other qualifying concern is that our ability, to date, effectively to relate runoff to water quality is at best inadequate. We are making strides in this direction and we are looking forward to the second generation of the 305(b) reports for better technical capability in this area.

Finally, this conference has re-emphasized, as Congress stated in its legislative history, that we know a lot about some of these problems and deal with them effectively. However, we still have to learn more about many of the issues and problems before we can develop effective programs for their control in any cost-effective manner.

PRIVATE CONTROL MEASURES
FOR NON-POINT WATER POLLUTION

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INTRODUCTION

Water-quality control has developed into a major function of government, and complex legislation and administrative machinery exist for regulation of potential sources of contamination. Although not as well developed as in the case of point discharges of pollutants, governmental controls increasingly are being applied to land-use activities that result in non-point discharges. But in addition to direct governmental regulation, control also is exerted by the private rights of individual citizens. Individuals may possess certain basic rights to uncontaminated water as members of the general public. However, attempts to assert such rights give rise to the problem of showing the existence of a legally protected interest and the standing-to-sue obstacle that is so prominent in environmental legislation.

A better defined, more enforceable right exists in the case of individuals who possess water rights in a natural source of supply. Water rights are a form of property which, in the eastern United States, usually exist in connection with the ownership of land in contact with a surface body of water or from which ground water can be withdrawn. Water rights have both a quantity and a quality dimension, and therefore provide a basis for private legal action to constrain contamination. The private right of action encompasses both the remedies of damage payments for injury already suffered and injunctive relief for threatened injury or that of a continuing nature.

Water rights do serve as a legal constraint on activities responsible for water-quality degradation. The fact that legislative and administrative controls have largely replaced the system of private rights as the primary institution for quality protection, however, indicates that there are serious problems with the effectiveness of private control measures as a management device. Of course, the basic weakness is that individual rights must be enforced by means of private lawsuits. The courts through which such rights are enforced do not initiate action; they become involved in a water-quality controversy only upon the motion of a party whose interests are sufficiently injured or threatened to initiate court action. This reliance on private initiation of action interjects an undesirable element of randomness into water-quality protection, since the water user injured by pollution may be reluctant to initiate court action for any one of a variety of reasons.

In addition to problems due to this discontinuous approach to water-quality management, the scope of judicial deliberations in deciding water-quality cases is generally too restricted in nature to allow development and implementation of a consistent public policy. Since the principal concern in a case under litigation is the settlement of specific damage claims or other narrowly focused issues, considerations of the public interest appear to be of secondary importance in a system based on adversary proceedings.

IMPACT OF GOVERNMENTAL CONTROLS ON WATER-QUALITY RIGHTS

Due to general recognition of these defects in private rights as a water-quality management institution and the widespread adoption of legislatively based programs of administrative controls, it is necessary to ask whether any role remains for private control measures. A basic factor bearing on this issue is the direct impact of statutory and administrative provisions on the legal status of private controls. The effect on individual rights may take one of three forms: (1) they may be strengthened or enhanced; (2) they may be restricted in scope, or (3) most commonly, they may simply be preserved by the provisions of the new controls.

Enhancement of Private Rights

Enhancement of the individual's rights occurs whenever his ability to initiate legal action is expanded or when his prospects for a favorable court decision are increased. A prominent example of the former type of enhancement is provided by the Federal Water Pollution Control Act Amendments of 1972. These amendments give any citizen the right to bring suit against any party in violation of certain terms of the amendments, or against the administrator of the Environmental Protection Agency to compel compliance with the mandatory provisions of the law [Federal Water Pollution Control Act Amendments of 1972]. This ability to initiate action to restrict waste-discharging activities without having to show specific injury to a water use is a significant expansion of the scope of private legal action.

Perhaps the best example of how legislative or administrative controls can enhance the individual's prospects of a successful lawsuit against a polluter is seen in situations in which strict liability is imposed for injury resulting from certain activities. Strict liability is a "no-fault" concept that provides for recovery of losses without proof of careless or improper conduct. The strict liability concept is therefore the theory of legal accountability most favorable to the injured party.

At the other end of the spectrum is the theory of negligence. The negligence concept is most restrictive from the viewpoint of the injured party, since liability is based on fault and therefore requires proof that conduct has fallen below some reasonable standard.

Both negligence and strict liability have been applied judicially to activities that result in water pollution. Whenever the establishment of legislative or administrative controls over certain activities results in a shift from negligence to strict liability as the basis for liability for related injury, the rights of the individual who may be adversely affected are significantly enhanced.

An example of such a shift in the basis for legal accountability concerning a non-point source of pollution is seen in the evolution of controls in the state of Texas with regard to handling and disposing of salt water resulting from petroleum extraction operations. The courts of Texas had traditionally refused to impose liability for water pollution resulting from the escape of salt water during such operations without specific proof of negligence, but an administrative rule adopted by the agency responsible for regulation of oil and gas production was interpreted in the case of Gulf Oil Corp. v. Alexander establishing strict liability in such cases.

Restriction of Private Rights

An example of how governmental controls can have the effect of restricting private rights is provided by the Virginia Erosion and Sediment Control Law. It provides that any party making a legal complaint in connection with injury from erosion, siltation, or sedimentation must show negligence in order to recover damages in connection with activities carried out in compliance with the requirements of the law. This provision precludes judicial acceptance of any theory of legal accountability, such as strict liability, that is more favorable to an injured party. Of course, any statute that limits private rights may be subject to constitutional challenge on the ground that it amounts to an uncompensated taking of property. The validity of this particular Virginia statute has not been tested in the courts.

Preservation of Private Rights in Existing Status

The third possible effect of governmental controls on private rights--preserving those rights in their existing status--is the most common. Preservation of the private right of action has been a central factor in certain cases where the validity of pollution-control legislation has been challenged. An example is provided by a 1965 Florida case, Hodges v. Buckeye Cellulose Corp. The landowner, Hodges, contested the constitutionality of a state statute allowing the discharge of municipal and industrial wastes into a stream near his property. The court held that the landowner did not have sufficient grounds to question the constitutionality of the statute, since the law did not create an exemption for injury related to such discharges and, for that reason, the landowner's rights remained unchanged.

The fact that governmental authorization of a waste discharge does not protect the discharger from liability for resulting contamination injury has been illustrated by specific court decisions. Payments for damages resulting from state-authorized discharges have been ordered, and the courts even have enjoined waste-disposal facilities specifically approved by a state pollution-control agency as in City of Marlin v. Criswell.

However, the fact that private rights are neither abolished nor explicitly modified by adoption of governmental control measures does not mean that such rights are unaffected. The court in a particular case may give considerable weight to agency determinations; in some instances, it may completely defer to agency judgment. The 1957 case of Ellison v. Rayonier Inc., decided in U.S. district court, is a good example. The case arose from the complaints of owners of coastal property in the state of Washington that an industrial waste discharge resulted in damage to oyster beds. Although the court acknowledged the continuing existence of private rights of action concerning water quality, it held that primary jurisdiction for pollution control had been vested in an administrative agency and that exercise of court jurisdiction should involve consideration of administrative determinations. The complaint before the court made no allegations as to the propriety of administrative action concerning the waste discharge. Therefore, the court assumed that the discharge was in compliance with all applicable controls and, on that basis, held that no recovery could be granted for violation of private rights. The court indicated that it would consider an allegation that the actions of the administrative agency had been arbitrary, capricious, or unreasonable, but in the absence of such a finding, relief for private injuries would not be granted.

In this case, the court adopted a rather restricted judicial position concerning the degree of water pollution to be allowed in a particular case. This position was that the trial of individual damage claims does not provide specific standards for resolving complicated technical and scientific problems, and that the scope of such proceedings does not allow adequate consideration of the public welfare. By its ruling, the court indicated that administrative procedures are the preferred mechanism for decision-making where the issues involve overlapping and conflicting private interests as well as the public welfare.

Note that this case was decided before the development of widespread interest and activism with regard to environmental quality. More recently, the courts have had to deal with an increasing amount of environmental litigation and have assumed a more active role in review of administrative decision-making. Nevertheless, compliance with governmental controls continues to be a factor often considered by the courts in resolving conflicts concerning water quality. Thus it can be concluded that, in most cases, the general effect of legislative and administrative water-quality controls is to reduce the significance of private controls existing in the form of individual rights, even where provision is made for their explicit preservation.

Due to this detrimental impact, it can be stated that the significance of private control measures is inversely proportional to the comprehensiveness and effectiveness of alternative governmental controls. Private controls have their greatest potential usefulness where direct governmental regulation is at a minimum. Because of the time lag and other problems in developing and implementing effective controls in the case of non-point sources of pollution, this area appears to be one in which a significant role remains for private control measures.

COURT DECISIONS INVOLVING NON-POINT POLLUTION

In determining the degree of constraint imposed on non-point sources of water pollution by private rights, a principal consideration is whether the particular activities that contribute to non-point pollution are "reasonable." In eastern water law, the concept of reasonableness is the judicially defined yardstick by which the legality of water uses or land uses affecting water is determined. Where water quality is involved, reasonableness is a relative concept. It requires a balancing of one party's right to uncontaminated water with another party's right to carry out a pollution-causing activity. Thus the nature of the source of contamination in a particular case is a basic consideration. A given level of contamination may be held to be an infringement of the injured party's

a landowner sought to enjoin an upper owner from allowing horses, cattle, and geese from entering and polluting the sources of a stream. The court refused to restrain the use of the water for what it saw as a natural use.

In contrast to these decisions refusing to restrain livestock pollution is an early case in the Supreme Court of California [People v. Elk River Mill and Lumber Co.]. That court did grant an injunction in a situation where cattle and hogs were confined upon the banks of a stream. The court indicated that a different decision would have been reached if the livestock had been confined at a reasonable distance from the stream. An injunction also was granted by the Supreme Court of Nebraska in the case of Barton v. Union Cattle Co., where wastes from a barn for feeding approximately 3,500 cattle were washed into a stream. The court acknowledged that the facilities were properly constructed and managed without negligence, but the magnitude of the operation and its impact on the stream in question were viewed as excessive.

In Atkinson v. Herington Cattle Co, a more recent decision by the Supreme Court of Kansas, damages were awarded to the owner of a water supply that had been contaminated by a feed-lot operation involving up to 7,500 cattle. In imposing accountability for the escape of contaminated water from the feed-lot, the court in effect applied the strict liability concept. The court gave no explicit recognition to this theory, but it cited as precedent an earlier Kansas decision, Berry v. Shell Petroleum Co., which had specifically endorsed the strict liability concept with regard to the escape of hazardous substances.

These decisions concerning pollution from animal wastes indicate that the courts are more likely to constrain pollution resulting from livestock operations as the scale and intensity of such operations increase. Where small-scale pasturing operations have been involved, emphasis has been placed on the "naturalness" of the activity, with no liability imposed for resulting pollution except in some cases of clear carelessness or negligence. In situations where larger numbers of livestock have been involved, especially under confined conditions, a stricter standard of accountability has been imposed.

Cases Involving Mine Drainage

Mine drainage is another area of non-point pollution where court interpretation of private rights has shown variation. One of the best-known decisions concerning pollution from mine drainage, and one of the most controversial, is Pennsylvania Coal Co. v. Sanderson. The case came before the Supreme Court of Pennsylvania on four occasions, with judgment entered in favor of the downstream landowner injured by mine drainage in the first three decisions. In the final deposition, however, the justices were divided in a four to three decision which denied the injured party's claim to damages. The court held that the mine owner had a right to make a natural use of his land, including the mining of coal. Since the discharge of mine water containing only natural impurities into natural drainage systems was viewed as an indispensable aspect of the mining operation, legal accountability was not imposed for the adverse consequences that resulted. The thrust of the decision was that private rights had to yield to the necessities of natural-resource development because of considerations of the public interest.

The Pennsylvania Coal Co. case has been frequently cited and extensively discussed by other courts. Defendants in pollution lawsuits often have relied on the case as precedent in support of their claims of immunity from liability. In general, however, the courts have not been receptive to the basic concept, and usually have decided that it is not applicable to the case under consideration or have specifically repudiated the concept. In a number of cases, the Pennsylvania finding has been rejected and liability imposed in connection with mine drainage. For example, the Supreme Court of Tennessee, in the case of H.B. Bowling Coal Co. v. Ruffner, where it upheld a lower court's award of damages for mine-drainage contamination, rejected the Pennsylvania case and stated that it is opposed by the great weight of authority in the United States. It should be noted, however, that most of the mine-drainage cases where liability has been imposed have involved situations approaching point discharges rather than seepage from wide areas.

Cases Involving Sediment Damage

Damages from sediment constitute a third area of non-point pollution where the courts have given some consideration to private rights. It appears that the majority of court decisions involving sediment problems have not involved water quality directly, but have concerned deposition of material on lands and in the channels of streams.

Few cases apparently have been decided concerning erosion from extensive areas of disturbed land. A substantial number of sedimentation cases have arisen in connection with mineral-resource extraction, where water used in the recovery and processing operations contained excessive sediment or where waste materials from mining operations were dumped into or near a stream. Resulting injury has frequently taken the form of flooding as the accumulation of solid material reduces the capacity of the stream channel to carry water. The courts generally have been inclined to grant legal relief for such injury [American Law Reports, 1953]. Although the physical situations giving rise to these cases approach point sources of pollution due to the concentrated nature of the responsibile activity, the same principles for establishing liability may have applicability to situations where sediment has more widespread sources, assuming that such sources can be adequately identified.

A few cases arising from sediment injury in connection with strip-mining operations are perhaps better representative of the non-point problem. One such case is a 1952 West Virginia decision, Oresta v. Romano Brothers, Inc. It involved the washing of disturbed overburden from an adjacent strip-mining operation. Three wooden barriers had been constructed to impound this material, but these efforts were not successful and deposition on the lower property reached a depth of three and one-half feet in some places. The court held that failure to confine the material to the mined land constituted negligence and imposed liability. A similar result was reached in the 1964 Kentucky case of Green v. Asker Coal Mining Co. The court imposed liability where erosion on strip-mined land resulted in deposition of soil, rock, and coal on the land of a lower owner, since no attempt had been made to restore the mined land.

Cases Related to Accidental Spills of Harmful Substances

The right of the individual to be free from water pollution resulting from the accidental escape of deleterious substances has received varying degrees of recognition in the courts. Reference has previously been made to the escape of salt water in connection with petroleum production. There is considerable acceptance of the strict liability concept with regard to salt-water injury, especially where the injury occurs on land not subject to a petroleum production lease.

Due to widespread and extensive storage and handling of gasoline and other petroleum products, a number of cases have arisen in connection with water pollution from these sources. The courts frequently have imposed strict liability for injury of this type [North Georgia Petroleum Co. v. Lewis, 1973; Yommer v. McKenzie, 1969; Swift and Co. v. Peoples Coal and Oil Co., 1936], but there is some variation among the states. Not all jurisdictions have accepted the strict liability concept. For example, the court in the 1934 Rhode Island case of Rose v. Socony Vacuum Corp., expressly rejected that concept and held that liability for contamination of ground water by petroleum products would not be imposed without a finding of fault. The Rhode Island court refused to make a landowner accountable for pollution resulting from a septic-tank failure without a showing of negligence. The strict liability concept was again specifically rejected.

THE EVIDENTIARY ISSUE IN NON-POINT POLLUTION CASES

In addition to the question of whether legal recognition is given to a water-quality right with regard to non-point sources of contamination, certain other issues also have an impact on the effectiveness of private rights as a constraint on contaminating activities. One of the most significant factors affecting the outcome of a pollution case is the evidentiary issue. As the plaintiff in a lawsuit, the injured party must overcome a substantial burden of proof, and failure to present sufficient evidence can be fatal to prospects for a favorable court decision.

One element of proof likely to be of special significance in cases involving non-point pollution is the establishment of a cause-and-effect relationship between the injury and a particular source for the contaminant. This problem is often a factor in pollution cases, especially were several potential sources of contamination exist. In some instances, suit may be brought against several contributors jointly, but the courts generally impose procedural limitations where the various activities responsible for pollution are independently managed and there is no concert of action.

In the case of non-point pollution, the problem of multiple sources is more likely to arise, due to the fact that the pollutants involved consist of such commonly occurring substances as sediment and animal wastes. In fact, in some cases, the pollution injury from non-point sources may be insignificant with regard to any individual contributor,although the combination of run-off from a particular watershed may have a substantial detrimental result. Thus, the evidentiary issue in many situations may be the principal determinant of whether private water-quality rights are enforceable as a constraint on non-point pollution.

CONCLUSION

The water rights of individuals have served and continue to serve as a constraint on water pollution from non-point sources. But it must be recognized that the effectiveness of this form of constraint is limited both by the inherent characteristics of the institutional arrangement through which water-quality rights are enforced and by the special nature of non-point pollution. The nature of certain sources of non-point pollution is such that the courts may view the polluting activity as a natural and reasonable use of land to which water-quality rights must yield, at least to some extent. In addition to the issue of whether the legal right to uncontaminated water will be enforced against activities responsible for non-point pollution, mechanical problems--such as proving the physical cause-and-effect relationship between the suspected source of a contaminant and the injury--are likely to pose a significant obstacle in situations involving non-point sources.

The extent of the deterrent effect which the existence of enforceable private rights has on activities that are potential sources of non-point pollution is largely determined by the prospect, as perceived by the potential polluter, that liability will be imposed for the adverse consequences of the particular activity. Because of the problems associated with the enforcement of water-quality rights in this area, there is a substantial probability that the individual contributor to non-point pollution can escape liability, a fact suggesting that the deterrent effect may be low. Thus the existence of private rights to uncontaminated water often may result in some curtailment of careless practices. It is unlikely, however, that substantial positive action to control discharges from non-point sources will be encouraged by the existence of private control measures alone.

This assessment of effectiveness of private controls lends support to the attempt to develop more direct and aggressive programs of governmental controls. Private controls will continue to function as an element of the over-all water quality management framework, but they are not well suited to function as a primary control mechanism. Their principal function will continue to be the equitable resolution of conflicts between individuals and the distribution of resulting losses, with any farther-reaching effect best viewed as a supplement to administrative efforts.

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AREAWIDE WASTE TREATMENT

MANAGEMENT OF NON-POINT POLLUTION

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INTRODUCTION

For those of you who have been attracted to this luncheon by the presence of Merna Hurd's name as the speaker, I wish to extend her regrets for not being able to attend the conference. Merna is the Program Administrator for the Areawide Waste Treatment Management Program now under way in New Castle County, Delaware, and I am the Project Director of the Program. Unfortunately, Ms. Hurd has been quite ill with a persistent virus, is under a doctor's orders to avoid travel, and has asked me to present my thoughts regarding the problems we have encountered in the evaluation of non-point pollution in our 208 Areawide Waste Treatment Management Program. Those of you who know of Merna, our 208 Program, and our separate personalities will know what I mean when I say you would probably have more enjoyed seeing and hearing Merna than you will me. In our organization, Merna is known for her ability with the rapier, while I am best known for my indiscriminate use of the bludgeon.

Within the book, When In Doubt, Mumble, a potential speaker is warned that in order to be successful he should always say what the audience wants to hear. In this regard, I may likely be unsuccessful today since I tend to look beyond the investigation of non-point pollution sources to the enforcement of justifiable controls in the public works arena.

My use of cynicism today, perhaps in excess, is done intentionally to convey to you my concern that we do something besides study non-point pollution. I never like to drive down a highway unless I have some idea of where I'm going, and I have participated in enough water-quality studies to recognize when the study itself becomes more important than what the study is to achieve.

Two experiences in my past may place my remarks today in context. In the first instance, I was an Army officer participating in a small-unit management course. During one case study, we were told that we were arriving at a site where we were digging a ditch. As we pulled up to the site, we were to notice that no one had shovels and everyone was sitting on the ground. The instructor then asked, "What is the problem?" When a student volunteered that the problem was obviously that the sergeant had forgotten the shovels, the instructor leaped upon the student's desk and screamed, "No! The problem is not a lack of shovels, the problem is digging the ditch."

The second experience is much more graphic. My major professor in my master's program once said to me, "Always keep your hands on the doughnut, but don't forget to look through the hole."

The title of this discussion as listed in the program was "The 208 Process: Where We Are and What We Need to Know." Since many of you here today may not be familiar with the numbers jargon of Federal programs, I have taken the liberty of changing the title of the address to "Areawide Waste Treatment Management of Non-Point Pollution." I would like to spend just a few minutes describing what a 208 Program is and what it isn't, and then devote the remainder of the address to the problems that I foresee in the evaluation of non-point pollution sources in a waste treatment management program.

THE 208 PROGRAM

First, with regard to the 208 Program, I feel an accurate definition of that process is necessary before I can place non-point pollution in the framework or perspective that we envision. Perhaps the most significant piece of federal environmental legislation to be passed in this century was the 1972 Amendments to the Federal Water Quality Act. These amendments, under Public Law 92-500, introduced such a multiplicity of programs and titles that most programs coined their name from the section number of that portion of the act applicable to the particular program. For those of us in the water resources field, program numbers such as 106, 303(e), and 208 have taken on the familiarity that we formerly reserved for our home phone numbers and, of course, the infamous 1040A.

You may wonder, if the amendments were passed in 1972, why it wasn't until 1974 that 208 planning programs were initiated. The answer apparently lies in the difficulty of initially defining the program. The introduction to Section 208 of PL 92-500 states that the program is applicable to areas which, as a result of urban and industrial concentrations or other factors, have substantial water-quality control problems. It is felt that the Environmental Protection Agency (EPA) should be made operational and placed under way before areawide waste treatment management planning was initiated. Under this theory, basic planning would identify significant problem areas that could only be investigated under a 208 program.

Now I still have not properly defined what a 208 program is and what it isn't. To understand the nature of the program it is necessary to realize that it is not an engineering analysis. The 208 program is a management process wherein a group of governmental entities enter into a joint alliance to investigate and solve complex water-quality and waste-treatment problems. The program combines the efforts of both planners and engineers, but it is foremost a management program. Since 208 is a management program and since the process includes representatives of all local jurisdiction, the investigation is implementation oriented. It is, therefore, imperative for a 208 waste-treatment management program to establish a reasonable balance between the long-studied point sources of pollution and the now fashionable non-point sources of water-resource contamination.

We in the New Castle County 208 Agency view the 208 program as an exciting opportunity, vehicle and mechanism for the actual implementation of a comprehensive water-quality management program. The 208 program, with its multi-jurisdictional framework, technical outputs, implementational requirements, and induced technical and citizen participation does not, to me, represent any drastically new concepts in the planning process. Rather, it is the enactment of a process that should have been the basis for all planning accomplished in the past. Thus, by legislation, we, the engineering and planning community, now have the opportunity to do what we have known has needed to be done for many years. If the 208 Program is to leave a mark on the United States, it is my hope that that mark manifests itself in empty shelves which previously have been reserved for completed planning studies.

NON-POINT POLLUTION IN PERSPECTIVE

In order for me to properly convey my apprehensions regarding the investigation of non-point sources of pollution, it is necessary for you to project yourself into the role of a public-works engineer charged with the responsibility of establishing and enforcing erosion and non-point pollution control standards. To establish enforceable standards, the public-works engineer must have legally defensible data. To the best of my knowledge, I don't believe that data does or will exist. In this regard, I am reminded of a short story by Carl Sandburg, who wrote of an encounter between an American Indian and a white man. Sandburg wrote:

> The white man drew a small circle in the sand and told the red man, "That is what the Indian knows," and drawing a big circle around the small one, "This is what the white man knows." The Indian took the stick and swept an immense ring around both circles: "This is where the white man and red man both know nothing."

To extend this analogy, I feel that the small circle represents the non-point controls the public-works engineer feels he can defend, the big circle represents what the scientist feels ought to be possible, and the immense ring represents where the lawyer feels that the engineer and the scientist know nothing.

The literature today is beginning to reflect the renewed interest that the engineering community has taken in non-point sources of pollution. One can read of the relative magnitudes of point and non-point loadings and be duly impressed that a substantial non-point pollution problem exists in this country. If the relative magnitudes are utilized to rationalize that tertiary treatment to remove an additional 10 percent of contaminant loads from a point source is not economical when the non-point loading exceeds the point loading by many orders of magnitude, then I can understand some purpose to the investigation. If the non-point investigation is conducted solely for the purpose of quanityring the non-point pollution problem, then I must state from a public works standpoint, "So what?" Does knowing that it is going to rain alter the weather?

This nation is now suffering through a severe recession, and public monies are not available to fund all the "do-gooder" engineering projects we may have envisioned that we needed five years ago. If we are faced with the allocation of a limited fiscal resource, then a system of priorities must be established for the disbursement of those funds. Construction of a wastewater collection and treatment system creates an identifiable point source of pollution, and responsibility then exists to ensure that the facility continues to produce an acceptable quality effluent. Perhaps it is too simplistic, but I prefer a philosophy of funding which lends priority to the alleviation of water-quality problems associated with prior-constructed facilities. In this regard, I would place a minimum of five wastewater priorities ahead of non-point pollution control. These five priorities would be: (1) the modification or upgrading of existing secondary facilities; (2) the expansion of secondary facilities; (3) the replacement of secondary facilities, (4) the construction of new secondary facilities, and (5) the installation of stormwater excess-flow clarification facilities at secondary facilities. I would, therefore, relegate non-point pollution control to a competitive position with tertiary treatment.

Harold P. Guy of the U.S. Geological Survey is a participant in this conference. In December of 1972 he prepared a brilliant paper entitled, "Urban Sedimentation--In Perspective," which was published in the Hydraulics Journal of the ASCE Proceedings. I cannot recommend it too highly as a document capable of keeping our feet in contact with the ground. We are now feeling the ground swell of the public's emotional concern regarding non-point pollution. This concern is no more evident than in concentrated urban-residential areas such as the arena in which our 208 program must function.

Guy noted that about 2.5 percent of the land area of this country is occupied by cities, 13 percent by agriculture, and about 83 percent by mountainous areas, forests, and marginal lands used basically for grazing and timber production. He notes that it has become fashionable to overlook the impact of agricultural cultivation on large areas and to become emotionally indignant at the sight of urban grading on relatively small areas. It is my opinion that, even though agricultural land use is intensified in the areas surrounding an urbanized complex, very little if anything will be done about the current level of agricultural pollution. Unless we are prepared to redefine the economic structure that establishes the profit margin for the farmer or for the forestry industry, we cannot expect to do more than gum to death the same principles the Soil Conservation Service has been trying to encourage farmers to adopt since the 1930's.

In urbanized areas, we are concerned with two significant sources of non-point pollution for which the remedies would face serious practical limitations. The first of these sources is the impact of a proliferation of septic tanks on the periphery of the urbanized area. Once again, unless we are prepared to redefine the economic climate which dictated the use of septic tanks, then a solution is not possible and study of the "septic-tank problem" is a useless exercise.

The second source of urban non-point pollution I will classify for the moment only as runoff, and let that classification include a multiplicity of possibly refineable problems. If we look beyond the study of urban runoff to the probable results, I think we can foresee the implementation problems that are at the heart of this address today.

I recently attended a Corps of Engineers public hearing regarding their ongoing study of the flooding problems on the major watershed within our 208 planning area. The hearing was held to promote public knowledge of local flooding situations. But as I sat there listening to citizens complain about water standing in low-lying areas, it struck me that the average citizen doesn't know the difference between flooding and poor drainage. For your convenience, let me separate the definitions as follows: Two to three inches of water in your yard or street is poor drainage; two to three feet of water in your living room is flooding. As we look toward possible solutions to the problem of urban runoff, only one economically viable methodology is promising--sedimentation by ponding.

Having examined the types of impoundments that can be and have been utilized to reduce sediment load, I am convinced that any possible non-point controls will have to be on-site rather than on the drainage system. Facilities placed within the drainage system generally have a design life such that their frequent overtopping or failure adds more to the problem than it adds to the solution. If this conviction is true, then the impoundments must be located on a multitude of off-system sites, perhaps in every neighborhood. If, as a public-works engineer, you order this done, you will satisfy that small percentage of the population concerned with non-point pollution. But you must bear the wrath of a substantial portion of the population who view check dams and impoundments as an eyesore that threatens the health and safety of their children who would naturally be attracted to play there.

Some of you may be concerned that my discussion has gravitated to one of sedimentation and erosion control and has ignored important runoff constituents such as nutrients, herbicides, pesticides, and heavy metals. Once again, assuming the role of the public works engineer, I must say, "So what?" So what if physical-chemical treatment promises the technology to rid our waters of these pollutants; so what do all the quantification and treatability studies mean if we can't do anything about the problem?

Non-point pollution is characteristically diverse both in contaminant concentration and spatial location. Realizing the importance of the time of concentration of the oft-touted first flush of pollutants, one can understand why any application of control measures must also be diverse in spatial location. If, then, the control measures must be diverse, the only facilities that have any practical application are those related to sedimentation and erosion control.

THE WASTE OF MATHEMATICAL MODELING

Within the areawide waste-treatment management community, I sense a desire to once again crank up long-dormant mathematical models, develop new modeling techniques, and attempt to collect a quantity of heretofore non-existent water quality data that will prove that, yes, we do have a very complex and diverse water quality problem on our hands. We in the engineering community have frequently been criticized for our apparent preference to study rather than act when faced with complex water-quality problems. Instead of searching for politically viable solutions, we have tended to give the study greater importance than the possible implementable results. We became so concerned with cost-effectiveness and environmental impact that we delayed for several years the upgrading of primary treatment plants to a level of secondary treatment.

We must bear in mind the objective of the non-point pollution mandate of Public Law 92-500. The Act requires that non-point sources be identified and that procedures and methods be set forth to control such sources to the extent feasible. From a public-works standpoint, I infer this to mean a system of uniformly applied, legally supportable implementation techniques and controls. The control methods adopted must be applied uniformly to be politically acceptable and must have some definable practical basis to be legally defensible, since the ultimate test of non-point pollution controls will be in the courtroom.

It is at this point that the pro-modeling community would press forward with their panacea models capable of rendering a very complex environmental problem into a neat stack of computer cards. My opposition to mathematical modeling of non-point wastes is rooted in the realization that engineering, and particularly its water-resources branch, is a very inexact science. If engineering is an inexact science, then non-point quantification is a gross science. Considering all the variables that affect a hydrologic phenomenon--for example, the number of variables in the Universal Soil Loss Equation--I feel that mathematical modeling of non-point wastes lends a degree of precision to the analysis that far exceeds that warranted by the problem.

Mathematical models are symptomatic of our professions's belief that it is more logical to make a gross number of small assumptions than it is to make a small number of gross assumptions. Our romance with models may be traced to a facination we all feel as we observe two opposing forces in competition with one another. Perhaps it would take a romantic to see in the Streeter-Phelps equation the eternal duality of nature and the struggle of good against evil. On the positive side, however, models did serve their purpose when they were applied to analysis of point discharges under some predetermined low-flow conditions. Under these conditions, some generally accepted parameters and techniques did establish themselves in the courtroom.

My concern regarding models is that they do not serve as a catharsis through which we purge ourselves of the desire to do something when faced with a complex engineering problem that defies solution. Engineering reports are noted for appendices of collected data that were never utilized in the report for making substantiative deductions or recommendations. The data, like modeling, frequently says, "Look how very hard we tried."

NON-POINT WASTE MANAGEMENT PLANNING

Two elements will be the key to the defense of a program of non-point pollution control. These elements are: (1) accurate supportive data, and (2) an acceptable methodology. At present, both elements are either non-existent or so controversial in nature that the program will not withstand an attack in court on a particular requirement placed upon an uncooperative developer or land owner. The first element, accurate data, does not--and probably will not--exist. Due to the general downstream location of most monitoring stations, it is probable that the water-quality data for a number of tributary land uses will be aggregated by the sampling technique. Thus in terms of a cause-effect relationship of any particular land use/soil type combination, supportive data will be weak. Further, if the variabilities encountered elsewhere are similar to those we have encountered in our area, no amount of frenzied sampling is going to improve the data situation in the time frame within which we must work. The second element, the methodology, will be as subject to controversy or difference of opinion as any water-resource evaluation. However, perhaps due to previous exposure to this element in other work, the problems encountered will probably be less unexpected.

With regard to soils disturbed by construction in the urban environment, we must remember that, as in military construction, the first principle of site work is to establish site drainage. This principle is an economic and a practical necessity if the contractor wishes to reestablish his work efforts as soon as possible after a rainfall. The principle, although economical and practical, is in direct conflict with what might be our initial perception of what sediment and erosion control ought to be. It is at this point that the public-works engineer must understand his responsibility to the public, to the contractor, and to the environment. In order to do that, he must accept the fact that erosion will occur during construction and he must plan for it. In this regard, the engineer must have a sediment-capture program and an erosion-control program. Prior to the initiation of site work, the contractor must be required to install his sediment-capture facilities. He will maintain these facilities until site work is complete and his erosion-control program--such as sodding, seeding, or grading--is established and approved. At that time, the contractor can remove the sediment-capture facilities.

CONCLUSION

Once again referring to the book, When In Doubt, Mumble, proper fulfilment of my role today as your speaker requires that I assist you in playing your role as an audience and warn you that the end of my talk is coming. The book recommends that I say, "In Conclusion," and then proceed to wrap up what I have to say.

Seriously, I would like to conclude with a brief summary of what we need to know in order to implement a program of non-point pollution control, a statement as to how I think we will obtain our objective, and in what direction I feel we ought to be devoting our attention.

First, we need tools of implementation, such as general relationships between land-use types and anticipated contaminant loadings, and we need a realistic analysis of control-technique feasibilities. At the present time, urban and rural non-point pollution data is similar to a black box; no signatures are worn by the contaminants, so we are unable to identify their source. Generalized relationships such as the ratio of fecal coliform to fecal streptococci bacteria, which indicate that wastes may be of human or animal origin, are desirable, but of little use in a management program. More supportative relationships are necessary, since what we have now probably is indefensible.

Second, I think the objective of a control program will be negotiated through public, private, and political cooperation rather than through a finite engineering analysis with a dictate of mandatory controls. In this negotiation, I think we can serve the public and the environment without unduly burdening the land developer or the community. A public works department can be a powerful tool in the enactment of measures it feels are in the public good. Its power probably can best be understood by the analogy to the cooks in the Army--you may complain about them and win a temporary battle, but you will surely lose the war. I feel, therefore, that if the control measures adopted are reasonable, then only the most foolhardy will attempt to challenge those regulations.

Finally, if at the conclusion of a multitude of non-point pollution investigations across the country we discover that what needs to be done is to utilize the "best management practice" uniformly and equitably by all sources, let us begin our search there--and devote our energies to the solution rather than to the problem.

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